Analysis of the Sonoma Developmental Center property for maintaining connectivity along the Sonoma Valley Wildlife Corridor: Implications for wildlife movement and climate change adaptation

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TABLE OF CONTENTS

SUMMARY	4
INTRODUCTION	6
Sonoma's mixed oak woodlands: A unique under-protected ecological community type	6
State and National Priorities	8
LANDSCAPE PERMEABILITY	10
Introduction: The importance of landscape permeability in corridor design	10
Methods: Landscape permeability model calculations	13
Distance to roads	14
Median patch size	15
Mean parcel size	16
Results	17
CLIMATE BENEFIT ANALYSIS	19
Quantified impacts of climate change in the Sonoma Valley Wildlife Corridor	22
Temperature: Winter (DJF) and summer (JJA)	23
Climatic diversity	24
Speed of climate change	25
Results	25
Temperature: Winter (DJF) and summer (JJA)	26
Climatic diversity	28
Speed of climate change	29
BUILT ENVIRONMENT ANALYSIS FOR SDC	30
Introduction	30
Methods	32
Results and discussion	33
MANAGING FOR CONNECTIVITY	34
Roads and traffic	35

Nighttime lights	36
Wildlife-friendly fencing	37
Domestic cat and dog presence	37
Recreation impacts	41
CONCLUSION	44
Landscape permeability	44
Climate benefit analysis	44
Built environment analysis	46
Managing for connectivity	47
MAPS AND ILLUSTRATIONS	50
1. Location map	50
2. Built environment impact envelope map	51
3. Landscape permeability map	52
4. Climate space map	53
REFERENCES	54

Summary

Habitat loss and fragmentation makes it difficult for animals to move through the landscape for daily activities and to disperse to new areas. Maintaining connections, or landscape corridors, between patches of habitat across the landscape can to allow separated populations to intermingle and breed, which can improve the persistence of species over the long term. As climate changes these landscape connections may also facilitate species shifts to more suitable climate conditions, and for this reason, habitat corridors, are one of the most common climate change adaptation strategies for biodiversity conservation. In an effort to determine where improving connections will make the biggest difference for species such as mountain lions, the Bay Area Open Space Council, identified Critical Linkages for the San Francisco Bay Area that including the Sonoma Valley Wildlife Corridor for important wildlife passage across southern Sonoma County. The importance of conserving the Sonoma Valley Wildlife Corridor to assist wildlife movement is in line with conservation objectives brought forth by state in the CDFW State Wildlife Action Plan and the Western Governor's Wildlife Council (WGWC).

The focus of the analysis in this report is on the potential for the Sonoma Valley Wildlife Corridor to allow for wildlife movement and climate change adaptation, as well as future management considerations to maintain and improve the habitat within the corridor. Barriers to connectivity in this region are associated with roads, buildings and human activity patterns. The research provided here includes estimates of landscape-scale permeability to help identify which natural areas have the least development and may provide safe passage for wildlife movement across the Sonoma Developmental Center property (SDC). There are advantages for maintaining this linkage between the Mayacamas Mountains and Sonoma Mountain for protecting species' access to a diverse range of climate types in an effort to increase the chances of adaptation under

pending climate change. There are also management considerations to maintain a functional wildlife corridor.

An important findings from the habitat connectivity analysis conducted is that the Sonoma Developmental Center property has high potential for landscape permeability and therefore is expected to allow for free passage of wildlife if left undisturbed; and represents one of the only options for wildlife movement between the Mayacamas Mountains and Sonoma Mountain that border Sonoma Valley. Historically, the difference between summer temperatures observed in the Sonoma Mountains as compared to the Mayacamas to the east is $2.7^{\circ} - 3.6^{\circ}$ F and the difference is to be $1.84 - 1.9^{\circ}$ F through 2099. Maintaining this corridor may be essential for some species in the region to adapt to climate change by shifting their distribution to cooler locations. The larger connected habitat patch that would result from conserving the Sonoma Valley Wildlife Corridor will also provide a greater overall diversity of climate types and that should be valuable for species adaptation in the future.

Protecting the Sonoma Valley Wildlife Corridor will require preventing further development especially in the northern portion of the SDC; as well as reduction in traffic speeds, artificial lighting, invasive species and domestic animal control, limiting human access, and a move toward wildlife friendly fencing throughout the corridor.

Introduction

Sonoma's mixed oak woodlands: A unique under-protected ecological community type

Located at the eastern edge of the coastal fog belt, the climate of the Sonoma Mountains and adjacent southern Mayacamas is intermediate between the cool, moist maritime conditions of the coast and the extremes of the more continental climate of the inland valleys. These factors have produced a rich flora and a diverse mix of vegetation types and plant communities including mixed conifer forest, mixed conifer-hardwood forest, oak woodland, mixed hardwood forest, grasslands, and a variety of riparian and other wetland habitat.

Perhaps no other plant reflects this biological diversity better than the oak (*Quercus*). The Sonoma Mountains support at least nine different species along with many undescribed hybrids. Large stands of Oregon Oak (*Quercus garryana var. garryana*) reach their southern-most limit in the Coast Ranges here, and together with Black Oak (*Q. kelloggii*), Coast Live Oak (*Q. agrifolia*), and Shreve Oak (*Q. parvula var. shrevei*) are common on wooded slopes. Other oaks found throughout the Sonoma Mountains include Blue oak (*Q. douglasii*), Valley Oak (*Q. lobata*), Interior Live Oak (*Q. wislizeni*), Canyon Live Oak (*Q. chrysolepis*), and Scrub Oak (*Q. berberidifolia*).

Due to the exceptionally high oak species diversity, this habitat type supports a myriad of birds and other wildlife. Our field studies across different housing densities throughout Northern California in these oak dominated landscapes document over 300 plant species and more than 80 bird species. Some of you are fortunate enough to know the thrill of spotting a Black-throated Gray Warbler, Warbling Vireo, or Wilson's Warbler; hearing a Downy Woodpecker; or gazing at Osprey and Red-shouldered Hawks above. While Sonoma Mountain still harbors a remnant of wilderness for residents to enjoy, there is much to do to protect these species for future generations.

Maintaining connected open space is clearly valued by the local conservation organizations such as Sonoma Land Trust, Sonoma Mountain Preservation, Sonoma Ecology Center, and the Sonoma Agricultural Preservation and Open Space District who have accomplished a good deal of habitat conservation through private land conservation tools such as conservation easements and acquisition of land.

Given that over 90% of California's oak woodlands are privately owned and state and local regulations do not generally prevent the clearing of oaks, private land conservation is essential if we want to maintain the biotic diversity supported by the oak dominated landscapes found in this region. Rapid rural residential and vineyard expansion threaten these diverse woodland communities. Studies show that these areas are not protected from exurban development. 73% of all of Sonoma County's remaining intact, natural forest could be comprised of edge habitat (within 500m of development) (Merenlender et al. 2005). The conversion of woodlands and forests is extensive in this part of Sonoma County. Converting oak woodlands to vineyards has discrete and identifiable effects, including the loss of vegetation cover, displacement of wildlife, soil disturbance, and habitat fragmentation (Garrison 2000). The California Department of Fish and Wildlife (CDFW) has listed 15 species that may be primarily affected by vineyard development in coastal California (Garrison 2000). Vineyard expansion is once again on the rise as a result of improved economic conditions and increases in global wine consumption, placing the Sonoma Valley Wildlife Corridor under extreme pressure for conversion to intensified agriculture as is observed in the surrounding hillsides and valley floor. In sum, the Sonoma Valley Wildlife Corridor and surrounding diverse plant and animal

communities are highly threatened by high value residential and agricultural development. Also, considerable costs have been expended to protect Sonoma Mountain to the west and the foothills of the southern Mayacamas to the east with the SDC presenting the most viable option for maintaining habitat connectivity across the valley floor.

State and National Priorities

Maintaining habitat connectivity and enhancing wildlife corridors is a cornerstone of California's State Wildlife Action Plan (Bunn et al. 2007). This plan mandates that "federal, state, and local agencies, along with nongovernmental conservation organizations, should work to protect ...wildlife corridors, and underprotected ecological community types." Wildlife corridors that offer significant benefits to underprotected ecological communities, and that are found in "areas where substantial development is projected", are a priority for state and federal land management and wildlife agencies to "protect from development those critical wildlife migration or dispersal corridors that cross ownership boundaries and county jurisdictions." The Sonoma Valley Wildlife Corridor crosses California's endemic oak woodlands, the majority of which are in private ownership. This corridor represents a unique opportunity to fulfill this important state mandate.

Projected climate change over the next few decades will change ecosystem structure, species composition, and diversity. Current climate change appears to be occurring substantially faster than in the pre-historical record, meaning that the ecological conditions required by many species (their niches) may be shifting faster than species can adapt. These pressures, caused by changes in climatic conditions encountered by species in their current distributions, are compounded by habitat loss and fragmentation. The resulting obstacles to migration may impede species' abilities to adapt to climate change to such an extent that many species could be

driven to extinction. Connectivity is one of the most commonly advocated strategies to help species adapt and survive the coming period of rapid climate change. The idea is that connectivity may allow species to shift their ranges in response to changing climate, and thereby allow evolutionary and ecological processes to be sustained.

Across the west, the value of conserving wildlife corridors has been recognized and large-scale corridor conservation efforts are being implemented. In fact, the Western Governors Association has an ongoing effort to assist with wildlife corridors and crucial habitat identification and conservation. It also recognizes the importance of understanding climate change impacts on wildlife corridors and crucial habitat, and the value of "taking steps accordingly to support adaptation to climate change (WGA 2008)." For this same reason, the goal of maintaining habitat connectivity for biodiversity conservation in California is prominent in the California Climate Adaptation Strategy where it's stated "to maintain natural corridors in anticipation of predicted climate changes should be factored into future local and regional habitat conservation planning efforts (CCCA 2009)." In particular, this strategic planning document encourages corridors that facilitate movement and incorporate temperature gradients that will benefit a suite of species. This has been our approach to the analysis of the Sonoma Valley Wildlife Corridor and SDC.

Landscape permeability

Given the existing development densities for the Sonoma Valley Wildlife Corridor area, we used existing models based on species assemblages to estimate the level of landscape permeability that remains. The landscape permeability models were derived from an estimated linear relationship between specific landscape features related to human land use (e.g. traffic volume, housing density) and bird and meso-carnivore detection levels from empirical field studies.

The permeability models were designed to make a general, community-level habitat quality assessment based on linear regression models derived from species assemblages in northern California (Merenlender 2011a). Gray et al. (in review) compared these biologicallyinformed, structural permeability models with animal field observations and showed that the model estimates do reflect animal habitat use on the ground. Thus, habitat permeability models constructed using information about animal response to human land use activities can be an informative component for land management and conservation planning in fragmented landscapes even when species data are unavailable.

Introduction: The importance of landscape permeability in corridor design

The pervasive spread of low-density development and resulting fragmentation continues to be an environmental issue of widespread importance and curtailing it presents a significant challenge for land use planners (Girvetz et al. 2008). The built environment, especially roads, urban and suburban development can reduce the ability for wildlife to move across the landscape (Fu et al. 2010; Tannier et al. 2012). Landscape permeability estimates offer a spatially explicit way to prioritize habitat connectivity for biodiversity conservation across fragmented landscapes (Gray et al. in review), which can be readily adopted by conservation and land use planners.

Landscape permeability estimates support planning where species information is unavailable. Permeability models may be the best approach to estimating or evaluating habitat connectivity when detailed animal location data is absent.

One of the primary threats to biodiversity is human-induced habitat fragmentation (Tilman et al. 2001; IUCN 2013), which is on the rise worldwide (Nilsson et al. 2005; Ribeiro et al. 2009; Butchart et al. 2010). A fragmented landscape is characterized by patches of natural habitat surrounded by a matrix of human-modified land cover (Mcintyre & Hobbs 1999). Protection of habitat connectivity is crucial for biodiversity conservation to facilitate movement through the matrix (Bennett 1999). Specifically, to conserve biodiversity we must identify and preserve core habitat patches supporting the persistence of species assemblages and ecosystems, and ensure connectivity among such patches with habitat linkages and/or a permeable matrix (Noss 2001; Crooks et al. 2011).

Increasingly, protected corridors are being planned and established to mitigate habitat fragmentation (Hilty et al. 2006) at multiple scales. For example, large-scale projects focusing on entire ecosystems are underway to connect forest communities from southern México into Panamá (Kaiser 2001) and linking the Yellowstone area in Wyoming north to Alaska (Walker and Craighead 1997). Similarly, local-scale projects to protect wildlife movement are happening worldwide (Underwood et al. 2011; Klar et al. 2012). Connectivity endeavors are often custom projects that depend upon species- and landscape-specific information (LaRue & Nielsen 2008), a practice that is expensive and time-consuming. Yet, land use and conservation planners often need connectivity assessment methods that can be rapidly developed and adapted into local and regional planning (Huber et al. 2012).

Connectivity metrics for biodiversity conservation differ in data requirements and informational yield. For example, structural connectivity is derived from landscape attributes such as the shape, size, and configuration of habitat patches, but does not account for animal dispersal ability. Structural connectivity estimates require less input data and generate relatively crude estimates of connectivity (Calabrese & Fagan 2004). Similarly, simple estimates of naturalness levels have been used to coarsely model landscape permeability across the entire United States (Theobald et al. 2012). On the other hand, functional connectivity is a measure of the ability of organisms to move among patches of suitable habitat in a fragmented landscape (Taylor et al. 1993; Fahrig 2003; Hilty et al. 2006). Ideally, measures of functional connectivity are derived from actual data about landscape composition, habitat use, and movement by wildlife. Such detailed data is uncommon at the landscape level because it is costly to collect. When empirical field data on species movement are unavailable, connectivity estimates can be derived from mathematical models. Models may be based on empirical studies of species' abundance or occurrence among different land cover types, or on expert opinion of species' habitat associations. Given the major influence a fragmented landscape has on connectivity among habitat fragments (Ricketts 2009), several models based on matrix connectivity have been developed including habitat resistance (friction; Ray et al. 2002; Joly et al. 2003), least-cost paths (Adriaensen et al. 2003), circuit theory (McRae et al. 2008), habitat permeability (Merenlender 2011b; Theobald et al. 2012), and linkage designs (Beier & Brost 2010).

Here we use landscape permeability models derived from an estimated statistical relationship between specific landscape features related to the built environment and species detections from empirical studies (Forman 2000; Reed 2007; Merenlender et al. 2009). Permeability models are an extension of the resistance concept (Ray et al. 2002); model output

often is in the form of a grid-based map with a value assigned to each cell that represents its permeability to an organism's movement. The permeability models were developed for linkage analysis by the Land Trust of Santa Cruz County (Merenlender 2011) and are designed to make biologically informed approximations of community assemblage responses to habitat quality (Metzger & Décamps 1997). The built environment--especially roads, urban and suburban development--can reduce the ability for wildlife to move across the landscape (Fu et al. 2010; Tannier et al. 2012).

Methods: Landscape permeability model calculations

We used regression models derived from meso-carnivore and bird assemblage response to human-modified land cover and landscape configuration as inputs to construct potential permeability maps. For each permeability map, we used as input a regression model derived from these two indices of habitat fragmentation: distance to roads (y_{ROADS} ; Forman 2000), median patch size (y_{PATCH} ; Reed 2007), and median parcel size (y_{PARCEZ} ; Merenlender et al. 2009).

Permeability model output is in the form of a grid-based map with a value assigned to each cell that represents its permeability to an organism's movement. We calculated each permeability map with ArcGIS 9.3.1 software (ESRI, Redlands, CA, USA). The geometric mean of the three regression models was calculated for each cell, and extrapolated across the study area to create the map of landscape permeability presented here (per Safner et al. 2011). All permeability values ranged between 0.0 - 1.0 with a cell size of 30 m x 30 m (900 m²). Permeability values are inversely proportional to habitat resistance or "cost"; a value of 0.0 indicates low landscape permeability, and a value of 1.0 indicates high permeability.

Distance to roads

There is overwhelming evidence of the effects of roads on natural communities (Fahrig & Rytwinski 2009), and thus we use distance from road, scaled by traffic volume (y_{ROADS}), as an index of animal response to transportation infrastructure. We calculated y_{ROADS} based on empirical data from several prior studies that evaluated the impact of roads on wildlife (Forman 2000; Reijnen et al. 1995, 1996; Forman & Deblinger 1998). Forman (2000) described the correlation between the distance to a road and bird species abundance and diversity. The closer a location is to a road, and the greater the road's traffic level, the larger the road effect, resulting in a corresponding decrease in abundance and diversity of birds that avoid urban areas. This approach assumes that the maximum magnitude of the road effect and effect-distance are proportional to the volume of traffic along the road.

We applied the equation derived by Forman (2000) to calculate the maximum effectdistance for each road in the study area as a function of mean traffic volume, measured as annual average daily traffic:

$$x_{ED} = 0.0126w_{TV} + 178.75,$$

where w_{TV} is the average traffic volume of the road, and x_{ED} is the road effect-distance.

We then assumed that the magnitude of effect of any given road would be proportional to the maximum effect and would decline linearly with increasing distance from the road. Thus, the road effect of each cell was calculated using the following equation:

$$y_{ROADS} = -\left(\frac{1}{\max(x_{ED})}\right) Z_{ROADS} + \frac{x_{ED} - \max(x_{ED})}{\max(x_{ED}) + 1},$$

where z_{ROADS} is the Euclidean distance from the nearest road and y_{ROADS} is the magnitude of the road effect.

We calculated the permeability map for y_{ROADS} with ArcGIS 9.3.1 software using road effect values from the equation for and the geographical position and orientation of all relevant landscape elements in the study area (per Safner et al. 2011). The traffic volume data came from the California Department of Transportation (<u>http://traffic-counts.dot.ca.gov</u>). In our study area, the maximum effect-distance max(x_{ED}) for all roads was 2812 m.

Median patch size

We used median patch size (y_{PATCH}) as a landscape-scale, area-informed index of habitat integrity calculated using the contiguity and relative size of proximate habitat patches. There is increasing recognition that area-informed metrics are useful to explain variation in wildlife abundances and movement capacity and perform well in analyses of landscape connectivity (Bender et al. 2003). We defined a patch as a contiguous area of habitat with natural vegetation cover and whose land use(s) were compatible with the establishment of mesocarnivore home ranges, based on information from prior space use studies. The model for y_{PATCH} was derived from a study (Reed 2007) investigating the correlation between patch size and mesocarnivore (e.g. coyote, bobcat, gray fox) occurrence in northern California, which found that the frequency of mesocarnivore detections increased with the size and contiguity of adjacent patches. yPATCH was calculated as the median area of habitat patches within a fixed buffer radius. In exploratory analyses, Reed (2007) found that y_{PATCH} measured at a buffer distance of 2,500 m explained the most variation in detections of the greatest number of mammalian carnivores. This work also revealed 'median patch size' to be a better predictor than buffered radius indices or proximity metrics (Reed 2007).

Per Reed (2007), we calculated y_{PATCH} using the equation:

$$y_{PATCH} = \frac{0.2356(x_{PATCH})^{\frac{1}{2}} + 1.385}{max(y_{PATCH})},$$

where x_{PATCH} is the median patch size in hectares (ha) within a 2,500 m radius buffer, and y_{PATCH} is the effect of habitat integrity on landscape resistance, measured as the density of native mesocarnivore detections along a survey transect.

As input data for y_{PATCH} , we used a map of terrestrial vegetation cover from existing land cover data (Farmland Mapping and Monitoring Program 2008) and removing roads (Research and Innovative Technology Administration, Bureau of Transportation Statistics 2001), mines and quarries, water bodies, and all land parcels less than 2 ha. We selected the larger patches in the landscape, which we defined to be any patch greater than 250 acres (101 ha). 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 km from any given point in the landscape – a range of median dispersal distances expected for terrestrial vertebrates found in the area. We used the equation for y_{PATCH} to calculate the patch size effect for each grid cell in the permeability map.

Mean parcel size

We used mean parcel size (y_{PARCEL}) as a local-scale index of human land-use intensity. Parcel maps may be 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). Empirically, prior research shows a substantial relationship between parcel sizes and some bird species and guilds (Merenlender et al. 2009). Specifically, Merenlender et

al. (2009) found that mean parcel size, calculated within a 500 m fixed radius buffer, was positively correlated with relative abundance of birds considered to be urban avoiders (e.g. Northern Flicker, Hutton's Vireo) in avian communities throughout the north coast region of California.

Per Merenender et al. (2009), we calculated y_{PARCEL} using the equation:

$$y_{PARCEL} = \frac{0.0211(x_{PARCEL})^{\frac{1}{3}} + 0.0155}{max(y_{PARCEL})}$$

where x_{PARCEL} is the mean parcel size in hectares (ha) within a 500m radius buffer, and y_{PARCEL} is the effect of parcel size on landscape permeability, measured as percent urban avoiding birds expected to be detected at any one location. As input data for y_{PARCEL} , we used a regional parcel map. We used the equation for y_{PARCEL} to calculate the parcel size effect for each grid cell in the permeability map.

Results

The landscape permeability model covered $3,688,200 \text{ m}^2$ across the SDC footprint, and was comprised of 4098 grid cells (900 m²). The distribution of permeability values for these 4098 cells ranged between 0.146 and 0.466 (Figure 1). Wildlife use of roads varies based on many factors such as animal type, body size, and mobility; and road width, composition, traffic volume, and traffic speed. Thus, a seemingly low permeability value of 0.146 as seen along Arnold Drive may not indicate that the road is a complete barrier to all varieties of birds or terrestrial animals.

Our results showed much of the northern portion of the SDC is of relatively high permeability. Specifically, 32% of the landscape had the highest permeability values – in a

narrow range of 0.43 - 0.466 (Figure 2). This distribution indicated that there is land of relatively high permeability within the SDC property, and such habitat is not rare. Further, 51% of the land in the SDC property had a permeability value between 0.35 - 0.5 (Figure 2), a more inclusive habitat permeability range that is preferentially used by wildlife, as demonstrated for pumas (*Puma concolor*) by Gray et al. (in review).



Figure 1. Landscape permeability map overlaid on the Sonoma Developmental Center footprint.

The distribution of the remaining 68% of the values was linear for low and intermediate permeability, indicating an even distribution habitat values between 0.146 - 0.43 (Figure 2). This linear distribution of values shows there was a mix of land quality across the SDC habitat with a similar amount of land with low and intermediate values.



Figure 2. (L) Distribution of landscape permeability values for the Sonoma Developmental Center. (R) Landscape permeability values for the Sonoma Developmental Center grouped into 0.05 unit bins.

By measuring landscape permeability associated with human development, this analysis offers a spatially explicit method to identify and prioritize habitat corridors for improved wildlife movement through the Sonoma Valley. While permeability data exists for the region beyond the boundaries of the SDC, we restricted this analysis to the habitat within the SDC footprint. Permeability at the SDC boundaries would be affected by neighboring landscapes and their use. For example, the presence of roads to the east and residential development to the south of the SDC would likely reduce landscape permeability, whereas the open habitat to the west would not. Expanding this landscape permeability analysis beyond the SDC to include the wider planning area would help us better understand the matrix within which the study area is situated. Lastly, we assumed all built structures are occupied and existing roads are in use within the SDC footprint. We would expect actual landscape permeability to be higher if some of the buildings are vacant or roads are unused. Additional analysis could include a revision of the model to incorporate current land use at the SDC.

Climate benefit analysis

Maintaining and improving habitat connectivity through the conservation of wildlife corridors or habitat corridors is the most frequently referenced tactic for increasing resilience of reserve networks to climate change. On the ground, this involves local efforts to prioritize small corridors across developed areas using parcel-scale data. A commonly used method for corridor planning uses a combination of species distribution and projected climate change models, both of which add a level of uncertainty to the output. Rather than basing long-term conservation efforts on a species-based approach, corridors can be designed based on the distribution and representation of climate space. For example, three ways the resilience of a reserve network to climate change may be improved are by prioritizing corridors that: 1) provide access to cooler climates, 2) maintain continuous habitat across a diversity of climate types, and 3) maintain access to areas with slower rates of change.

Landscape corridors allow for adaptation to climate change

The pressures caused by changes in climatic conditions encountered by species in their current distributions are compounded by habitat loss and fragmentation, resulting in potential barriers to migration that may impede species' abilities to adapt to climate change to such an extent that many could be driven to extirpation or extinction. Habitat connectivity is one of the most frequently promoted strategies to help species adapt to rapid climate change resulting from anthropogenic disturbance (Heller & Zavaleta 2009), and for the same reason habitat corridors have been adopted to make protected area networks more resilient to climate change (Hilty et al. 2012).

Much of the climate change analysis for habitat connectivity planning is done on a continental or global scale where global climate data is used to infer shifts in species distributions based on the velocity of change (Burrows et al. 2014) or to track shifting habitat suitability (Lawler et al. 2013). However, when it comes to implementing even the most grandiose corridor plans, local conservation organizations and stakeholders rely on fine scale

data and personal knowledge to prioritize land protection and management strategies (Hilty et al. 2012). This type of on-the-ground connectivity conservation requires prioritization of small corridors across developed areas.

The most common approach to incorporating climate change scenarios into habitat connectivity planning is to track how a species' climatic envelope (suitable temperature and moisture regime) changes across a landscape under future climate scenarios. A corridor is then delineated to facilitate movement from the current species distribution to areas predicted to be more suitable in the future (Lawler et al. 2013). This approach, while intuitive, combines species distribution models – with high levels of uncertainty due to the limited understanding and use of species biology – with climate change models that have a wide range of outcomes depending on future levels of greenhouse gas emissions as well as how the atmosphere and oceans respond to these emissions. In addition, species climatic envelope predictions often rely on extrapolating modeled conditions based on species' reliance on current climate condition into different future climate scenarios for which we have no data to support or deny the aptness of these novel climates for individual species persistence.

A simpler alternative, which avoids the inherent uncertainties in a species-based approach, is to design corridors based on the expected rates of climate change and the distribution of climates across space and time. "Climate space" is one way to express the range in temperature and precipitation regimes that exist in a location. Nuñez et al. (2013) prioritizes pathways that maintain climate stability by minimizing the slope (change) of climate within a corridor, then selecting corridors between reserves that follow the lowest cost path, as measured by the smallest climate differences. Here we consider climate stability as one way to identify the priority corridors to protect across a landscape. We also consider the advantages of climate

diversity that Loarie et al. (2009) point to for protected areas, as well as corridors that would facilitate movement to cooler climates. All three of these factors – climate stability, climate diversity, and access to cooler climates – have been shown to influence the resilience of a reserve network to climate change (Merenlender et al. in prep.). A comparison of these three approaches is based on the following assumptions: 1) maintaining access to cooler climates is a high priority, 2) a reserve network that harbors greater climate space diversity will allow for greater adaptation, and 3) slowing the rate of climate change will provide the greatest advantage for species trying to adapt or relocate.

Quantified impacts of climate change in the Sonoma Valley Wildlife Corridor

To illustrate the differences among climate space metrics for prioritizing corridors we used existing analyses of climate space (Merenlender et al. in prep.) to quantify the current and future climate diversity and temperature gradient that this Bay Area Critical Linkage provides. To calculate the value added by maintaining the Sonoma Valley Wildlife Corridor, we examined different characteristics of climate in the corridor based on three distinct assumptions for improving reserve network resilience to climate change: 1) access to cooler climates (temperature); 2) maintaining continuous habitat across a diversity of climate types (climatic diversity); and 3) maintaining access to areas with slower rates of change (speed of climate change).

We assumed that a corridor will enable species to access neighboring patches, and without which the species are restricted to the climate space within a patch. This assumption that the developed matrix between habitat patches prevents species movement is a common one in habitat connectivity analysis (Hilty et al. 2006). We defined a corridor as a segment of land connecting two or more patches of permanent habitat. While a corridor may support wildlife, the

purpose of our analysis was to evaluate how increasing connectivity affects the patch network. As a connector or thoroughfare, we did not consider a corridor to be suitable for permanent habitat, so climate benefits were only realized by adding a patch. Consequently, we did not consider the values within the corridor when calculating the final benefit of connecting two patches. This information was based on recent analysis for the Mayacamas and surrounding areas.

For the Mayacamas study, all historical climate information came from PRISM (Para meter-elevation Relationships on Independent Slopes Model), an interpolation method that describes spatial climate patterns in the United States (http://www.prism.oregonstate.edu/) (Daly et al. 2008). The 4 -km resolution digital elevation model in PRISM was used prior to bias correction for spatial downscaling (Flint & Flint 2012). A recently developed Community Climate System Model (version 4.0; CCSM4_rcp8.5) global climate change model, Community Climate System Model (version 4.0; CCSM4_rcp8.5), was used to assess changes in climate space over time. Thirty-year averages were used and spanned the following time intervals: 1951-1980; 1981-2010; 2010-2039; 2040-2069; and 2070-2099.

Here, we calculated the three climate space metrics for the two corridors that overlapped with the SDC as well as the two adjoining patches joined for each 30-year period with the exception of velocity, which returns the speed of change between historical averages and 2070-2099 averages (Figure 3).

Temperature: Winter (DJF) and summer (JJA)

Access to warmer habitat during cool winter months, and to cooler habitat during warm summer months, is important for mobile animals in the immediate term and dispersing plants and animals in the long term, particularly in light of changing climates. We calculated the difference

between the lowest patch grid cell values for winter minimum temperatures (average of December, January, and February means; DJF), and assigned this value to the corridor linking the two patches to represent the added benefit of the network in maintaining cooler winter minimum temperatures. Similarly, we calculated the difference between the lowest patch grid cell values for summer maximum temperatures (average of June, July, and August means; JJA) to represent the added benefit of the network in maintaining cooler summer maximum temperatures.

Climatic diversity

Climatic water deficit (CWD) quantifies evaporative demand exceeding available soil moisture, and is used to estimate measures of soil moisture and climate (Stephenson 1998). As a calculation of the amount of water (in millimeters) by which potential evapotranspiration exceeds actual evapotranspiration, CWD is a proxy for how plants experience and respond to climate change. Recent studies suggest CWD may serve as an effective control on vegetation cover types in the San Francisco Bay Area and is believed to be especially predictive in Mediterranean-climate regions, due to the long dry season these communities must sustain.

We calculated the diversity of CWD as described by the Rao equation presented in Ackerly et al. 2010 using as input the values for CWD across all five 30-year time intervals. Here Rao's quadratic entropy (Rao 1984) was modified for a continuous distribution, and incorporated evenness and degree of spread for the CWD values; where d_{i,j} is the absolute difference in CWD between grid cell values i and j within a patch and N is the number of grid cells that fall within the patch.

$$\mathbf{S} = \frac{\sum_{i=1}^{N-1} \sum_{j=i+1}^{N} di, j}{\frac{1}{N^2}}$$

We calculated one Rao value for each patch derived from all grid cells therein. Then we took the absolute difference between the Rao calculations for each connected patch, and assigned the difference to the adjoining corridor. This value represented the increased amount of CWD diversity the network presented over any one patch.

Speed of climate change

To calculate the difference in velocity of climate change, we used methods described in Loarie et al. (2009), based on average annual temperature across 1981-2010 and 2070-2099. The resolution of our climate data was finer (270m) than that used by Loarie et al. (2009), and our future temperatures were estimated based on CCSM4_rcp 8.5. To find the velocity of change in km/year for each grid cell, we calculated the historic temperature spatial gradient (% slope), divided the slope values for each patch by the difference in mean average annual temperature between historical records and future estimates, and then multiplied by the number of years between these data sets (29 years). We determined the extent to which the network of grid cells offered habitat with lower velocity values by calculating the absolute difference between the lowest velocity values in each patch and attributing this value to the connecting corridor.

Results

The SDC overlaps with patch p534 and corridor c632, and is thus part of a key linkage between two large patches of undeveloped habitat on either side of the Sonoma Valley (p534 and p474; Figure 3). These two patches represent relatively large regions of geographic and topographic diversity that could otherwise be separated by residential development in the area. Additionally, protecting the SDC would widen the proposed corridor c632, offering additional connectivity in this bottleneck between the habitat patches.

Additional climate change analysis for this region could include a reevaluation of patch delineation on a smaller scale. The results we present here were calculated using existing large-scale data due to time constraints.



Figure 3. Map showing the geographic configuration between patches p534 and p474, corridor c632, and the Sonoma Developmental Corridor.

Temperature: Winter (DJF) and summer (JJA)

Corridor c632 provided a greater advantage for facilitating access to cooler summer temperatures than cooler winter temperatures. Based on the DJF temperature model calculation, corridor c632 offered between 0.11 - 0.12 °C climate advantage during winter minimum temperatures from 2010 – 2099. While this may seem like a small amount of climate benefit, this is within the temperature range predicted by the winter model for 30% of the corridors throughout Northern California (Merenlender et al., in prep.). One reason the temperature

advantage is greater in the summer than in the winter is because the maximum temperature variation in Northern California is found during the summer, when severe differences may be found between the marine dominated coastal area and interior areas. In summer, average maximum temperatures are 14°C cooler along the more coastal ranges than inland as compared to 1°C warmer along the coast than inland for average winter minimum temperatures.

In the JJA temperature model, corridor c632 offered between 0.94 - 1.06 °C cooling during warm summer months over the next 85 years. A climate advantage greater than 1°C for the JJA model was observed in only 41% of the 794 corridors examined by Merenlender et al. (in prep.) across the Mayacamas Mountain region in California. This intermediate level of climate advantage offered by the corridor during summer could be related to the amount of topographic and geographic diversity offered by p534 and p474. Further, p534 is much larger than p474, and when corridors connect two patches of disparate size, a greater climate benefit is realized for the smaller of the two.

Corridor c632 provided a greater advantage for facilitating access to cooler summer temperatures than cooler winter temperatures. This is because the maximum temperature variation in this region is found during the summer, when severe differences may be found



Figure 4. Climate benefit offered by corridor c632 during winter (DJF) and summer (JJA) months across 5 time periods.

between the marine dominated coastal area and interior areas. In summer, average maximum temperatures are 14°C cooler along the more coastal ranges than inland as compared to 1°C warmer along the coast than inland for average winter minimum temperatures. Temperatures have dropped to as low as 14°C at the highest elevation areas, but usually range from 15°C to 20°C throughout the central part of the study area. Many peaks in the neighboring hills and mountains connected by c632 are around 500 m; resulting in temperature differences due to change in elevation of approximately 5°C across the steepest terrain.

Climatic diversity

Based on the cumulative water deficit (CWD) model calculation, corridor c632 offered between 42.75 - 44.47 units of climatic diversity advantage between 2010 and 2099. These levels of climate diversity advantage are high in comparison to the values predicted for 794 such corridors throughout Sonoma and surrounding Counties. Specifically, over the same time period the median climate diversity benefit calculated by Merenlender et al. (in prep.) was 30.17 -31.12, and the 75^{th} percentile value was 43.38 - 44.49. It is also worth noting that the amount of climate diversity provided by corridor c632 was predicted to increase over time.

By providing additional climate diversity over the next century, the land at the SDC site will be of increasing value in the face of predicted climate change. Diversity of CWD may have some value for ensuring the maintenance of high levels of plant community diversity; however, just how much more species diversity likely results from an individual corridor is hard to predict. High rates of CWD diversity are related to topographic diversity and habitat patch size. The largest changes in overall CWD diversity occur when small isolated patches of habitat are connected to large, more topographically diverse patches. If increasing the diversity of moisture regimes for plant species persistence is a priority we would argue that corridors should be a priority for the more fragmented part of the study area where urban and agricultural development have resulted in smaller remnant habitat patches that contain less topographic diversity than the more northern larger patches.

Speed of climate change

Based on the speed of climate change model calculation, corridor c632 offered a reduction in the velocity of climate change of 0.11 km/year between historical averages and 2070-2099 averages. While this may seem like a small amount of climate benefit, this is within the temperature range predicted by the winter model for 25% of the corridors throughout the Sonoma County region (Merenlender et al., in prep.). For example, velocity grid cell values ranged from 0-24 km/year for all of California. Most grid cells in the region surrounding the SDC had velocity values of lower than 0.1 km/year, as was also observed for this region in a previous statewide analysis at a coarser scale (Loarie et al. 2009).





Figure 5. Cumulative water deficit (CWD) diversity offered by corridor c632 across 5 time periods.

climate change as fast as other less topographically diverse parts of California. Hence, very little difference exists between the minimum velocity values for the two patches we examined, making targeting for slower climate change less useful than it could be for larger landscape corridors. With greater climate stability across the region comes opportunity for conservation of biodiversity refugia emphasizing the importance of protecting large continuous wild lands for California's Mediterranean-climate adapted species to persist over the next 100 years.

Built environment analysis for SDC

Introduction

The most important ecological benefit of the SDC property is to provide habitat connectivity across the Sonoma Valley Wildlife Corridor, which has been impacted by habitat loss and fragmentation due to an increase in vineyard planting and exurban development. Rural development has enormous potential to fragment the remaining wildlands that provide refugia for wildlife, community separators, and open space amenities. Habitat fragmentation is considered by many scientists to be the largest threat to preserving the world's biodiversity and the major cause of extinction today (Henle et al. 2004). The biological consequences of habitat fragmentation range from a decline in numbers of species, population sizes, contracted ranges, and increases in exotic species (Beier 1993; Wiens 1996; Stefan 1999). Part of the problem is that fragmentation increases "edge habitat" that impacts biodiversity and ecosystem function. The division of one continuous natural habitat by humans into one or more smaller remaining fragments of habitat results in a human-created edge where the natural habitat ends and abuts the human-altered parts of the landscape. The hard-edged boundaries that often result from human

disturbance have a stronger negative impact compared to more natural transitional edges (Mesquita et al. 1999).

There are both physical and biological consequences associated with edges (Ahern 1995; Laurance et al. 2002). Such influences can extend as much as 1,500 feet into forest patches (Laurance 1997; Sizer and Tanner 1999). These altered conditions can inhibit regeneration of vegetation where seeds are particularly sensitive to desiccation and can increase mortality due to trees being uprooted or broken by the wind (Laurance 1997). For very small fragments of natural habitat, the entire patch may be affected by these micro-climatic changes associated with the edge. Such changes in micro-habitat and consequently to natural vegetation can be one of the contributing reasons for corresponding faunal changes in composition and density.

Generalist predators and exotic species often prefer edge habitat and can contribute to a negative edge effect by out-competing specialists and native species and can also result in increased predation on native fauna (Beier 1993; Wiens 1996; Stefan 1999). Because of large edge-to-area ratios, smaller habitat fragments with higher edge to area ratios provide increased access of weedy species into fragments and can enhance movement of edge-loving exotic species and pests (Panetta and Hopkins 1991). Brown-headed cowbirds (*Molothrus ater*), raccoons (*Procyon lotor*), opossums (*Didelphis virginiana*), and crows (*Corvus* spp.) are examples of species that thrive in edge habitat and can have a large impact on forest interior species. Such species act as nest predators, nest parasites, or cavity competitors of interior species, and they can contribute to decreased populations of ground-nesting birds, forest songbirds, reptiles, and amphibians in remaining habitat fragments (Harris et al. 1996; Dijak and Thompson 2000; Hansen et al. 2002).

Species that may spend most of their time in the human-impacted regions can also impact biodiversity by invading forest edges and smaller fragments (Stefan 1999). Domestic and feral animals, such as cats and dogs, which come from human dominated landscapes, can damage native species populations in remaining habitat by chasing and preying upon them (Arango-Velez and Kattan 1997; Crooks and Soulé 1999).

To recognize the ecological problems associated with habitat fragmentation by the built environment across the SDC property, we mapped buildings and roads to visualize and estimate their influence on wildlife. The assumptions used here about the intensity of existing roads and occupancy of structures may overestimate the impact of the built environment at the site at this time because some of the buildings are not currently in use and we have not identified studies of how abandoned buildings influence wildlife abundance.

Methods

To calculate the area impacted by the existing structures on the SDC site, we applied a fixed impact buffer of 30 meters that encompassed the large trails, roads, and 231 buildings within the SDC footprint. Physical and biological impacts on a wide variety of life forms – including trees, understory birds, mammals, amphibians, and various invertebrate groups – have been detectable as far as 1,640 feet into forested systems (Laurance 1995). However, a 30 meter impact zone around the buildings, roads, and large trails was used because there is strong evidence that the abundance of native birds that are not urban adapters drop precipitously within 30 meters of rural residential structures (Odell and Knight 2001).

Results and discussion

The area of the SDC property that is being proposed for increased protection from development is part of one of the largest core mixed oak woodland³ entirely within Southern Sonoma County (13,970 acres). The majority of the buildings (n=172) are clustered in the center of the SDC footprint, along the southern border of the property impacting an estimated 7954.52 square feet or 182.82 acres. There is a smaller cluster of (n=41) buildings in the eastern flank of the property impacting an estimated 699.65 square feet or 16.18 acres. This clumped distribution of buildings aggregates the impact of the built environment into two primary regions within the SDC, with the remainder of the property relatively unaffected by buildings and roads. As a result, the density of the buildings adjacent to a cluster of roads at the center of the SDC renders this portion of the site relatively impermeable. High building density effectively creates a bottleneck for wildlife movement along the northern border of the property that is at most 689 feet wide and 2560 feet long. Given that wildlife may avoid the parts of the landscape identified by the built environment buffer, the width of the bottleneck could shrink to between 130 – 420 feet when buffered land is subtracted from the overall bottleneck footprint.

Strong differences in species composition are expected in the developed areas mapped in Figure 2. A study done in Sonoma County illustrated the impacts of subdividing oak woodlands to native biological diversity (Merenlender 1998). The project compared relatively undisturbed oak woodlands (greater than 500 acres) to ranchettes of 10 to 100 acres, and to single-family homes on lots between ¼ to 2.5 acres. This research suggests that rural development in these areas will support more birds adapted to urban conditions and a greater degree of exotic plants

³ Core oak woodlands in the Sonoma County Agricultural Preservation and Open Space District Acquisition Plan 2000 are defined as large (> 50 acres) continuous interior hardwood-dominated communities identified from the California Department of Forestry (CDF) vegetation map, which is based on 1990 satellite imagery with 100-foot by 100-foot resolution. All core oak woodlands included in the oak woodland priority map were below 1,700 feet because these low elevation areas were considered to be more susceptible to development.

that have less ecological value to native insects and vertebrates that the areas found in the northeastern part of the SDC property. In a wildlife camera study through varying densities and configurations of housing development, Goad et al. (2014) showed how the impacts of exurban development on mammals are species specific and vary along a development gradient. At the SDC property, it is likely that some mammals like red foxes (*Vulpes vulpes*) would respond positively to development. However, many small- and medium-sized mammals, including bobcats (*Lynx rufus*) and coyotes (*Canis latrans*), could decline or disappear as development levels intensify.

Future work could include field data collection that would greatly improve our understanding of the impacts of the mapped buildings on wildlife abundance as compared with less developed areas. Areas adjacent to buildings with different levels of use could be surveyed to examine their habitat suitability for species of concern Some buildings may be frequently visited by large numbers of people and, at the other extreme, other buildings may be vacant with no regular human presence. The influence of relative building use could be integrated into the built environment analysis to give a more detailed description of land use across the SDC footprint. Removing isolated buildings and any not required for future use is highly recommended to enhance wildlife movement and the overall ecological integrity of the SDC property.

Managing for connectivity

We reviewed the scientific literature and report here the current knowledge about the impacts of traffic speed, nighttime lights, domestic dog and cat presence, fencing, and recreational land use impacts on wildlife. These sources address impacts on birds and mammals in terrestrial systems within temperate regions (i.e., no snow-related impacts). The information

presented below is a summary of results of previously published studies that were conducted at locations outside the SDC. An assessment of some or all of these management factors at the SDC could support the recommendations made in this report, as well as contribute to a greater scientific understanding about habitat management and conservation.

Roads and traffic

Road ecology is a relatively new field, with steady growth in the number of journal articles, books, conferences, and "best practice" guidelines since the publication of *Road ecology: science and solutions* (Forman et al.) in 2003. To investigate the concern that roads and traffic may be reducing or eliminating wildlife populations, Fahrig and Rytwinski (2009) reviewed the published literature on road ecology and synthesized their findings. In their review, Fahrig and Rytwinski (2009) showed that in results from 79 studies, covering 131 species and 30 species groups, negative effects of roads on animal abundance outnumbered the positive effects by a factor of 5 (114 negative, 22 positive, 56 no effect).

One way to improve road safety and mitigate the negative impacts of roads and traffic on wildlife is traffic calming (Jaarsma et al. 2013). Traffic calming reduces traffic volumes and speeds on minor roads at a regional scale and can be implemented with speed reducing devices and planning traffic routes such that main traffic is directed onto major roads with higher speeds while lower volume local traffic uses smaller roads with reduced speeds. Models investigating the effects of traffic calming on wildlife mortality have been shown to increase the persistence of roe deer in a landscape with a dense road network (van Langevelde and Jaarsma 2009). The SDC likely experiences inadvertent benefits of traffic calming, as the speed limit through the property is 15 - 25 miles per hour, and has historically been well enforced.

Wildlife crossing structures can also facilitate animal movement across roads. Crossing structures function best when designed for the animals that will use them. Large overpasses that span roads and freeways are successful in helping large mammals like grizzly (*Ursus arctos*) and black bears (*Ursus americanus*) cross highways (Sawaya et al. 2013), whereas culverts and below-road passages are sufficient for animals of a variety of body sizes including coyotes (*Canis latrans*) and bobcats (*Lynx rufus*) (Alonso et al. 2014), as well as pumas (*Puma concolor*) (Gloyne and Clevenger 2001).

Nighttime lights

There is growing evidence showing the negative impacts of artificial night lighting across numerous wildlife taxa, which has also been identified as a key biodiversity threat (Hölker et al. 2010).

Artificial lighting been shown to alter individual bird and animal behavior, reproductive success, and survivorship (Longcore & Rich 2004). For small, nocturnal, herbivorous mammals, artificial lighting can greatly disrupt foraging behavior and increase predation risk (Kramer & Birney 2001). Constant lighting has been shown to modify an individual's circadian rhythm and melatonin production in nocturnal mammals (Sharma et al. 1997); alter reproductive success across a wide range of taxa (cottontail rabbits: Bissonnette & Csech 1938; green frogs: Baker & Richardson 2006; blue tits: Longcore 2010); increase the incidence of ungulate road kill (Beier 2005); and interfere with dispersal movements and corridor use by larger mammals like the puma (Beier et al. 1995). Further, nocturnal lighting has been shown to alter higher levels of biological organization beyond the individual. Changes in community composition due to artificial lighting (Davies 2012) may ultimately alter ecological structure and function. To

mitigate the negative impacts of artificial nighttime lighting at the SDC, the number of active lighting fixtures and the intensity of their bulbs could be decreased.

Wildlife-friendly fencing

Fences visibly and physically delimit property. Functionally, fences control access to land by humans and animals. For example, fences allow livestock or wildlife to be confined to particular landscape patches, which can exclude herbivory, control erosion, and protect waterways (Boone & Hobbs 2004). Standard perimeter fencing can also negatively impact wildlife by creating a barrier to local movement and seasonal migration. Additionally, improper fence design can result in animal injury or death as a result of collision or entanglement. A wide spectrum of animals may be injured by fencing -- from ungulates whose hoofs can be caught in barbed wire, to waterfowl like swans and blue herons that can be ensnared by fences that block flyways (JHWF 2013). Landowners must then face the undesirable work of clearing the animal carcass from the fence and paying for costly fence repair.

In contrast, fencing that is considered "wildlife friendly" allows free passage of wildlife and increases visibility to prevent animal ensnarement and mortality. Thus wildlife friendly fencing improves habitat and provides better access to water, food and shade. Guidelines for fencing that is considered "wildlife friendly" are publically available online and from local Land Trusts, Resource Conservation Districts and other natural resource agencies. .

Domestic cat and dog presence

Free-roaming and feral domestic cats (*Felis catus*) are the most significant exotic predators worldwide. Cats have been introduced on six continents, are able to exploit a wide range of habitat types and prey species, and have high rates of population growth. The U. S.

population has doubled since 1970 and a recent estimate includes 66 million domestic pets and 60-100 million stray and feral animals (ABC 2002; Nassar & Mosier 1991); such unregulated populations pose significant threats to native species and biodiversity. To the extent that domestic cats are generalist predators, subsidized by pet owners and animal welfare groups in backyards and protected areas alike, habitat suitability is likely to impose few limits on their population expansion.

Adverse impacts of free-roaming cat populations on prey species are well documented. For example, a study in Wisconsin showed that cats are responsible for killing as many as 217 million birds annually in that state (Coleman and Temple 1995). Cats are credited with eight extinctions and 40 extirpations of birds in island systems in New Zealand (ABC 2002), and the presence of cats has been shown to be the most important factor in the extinction of native mammal species in many Australian islands (Burbidge & Manly 2002). In San Francisco Bay Area regional parks, Hawkins (1998) demonstrated that the presence of cat colonies correlated not only with reductions in prey densities, but also a significant shift in prey composition from native to exotic species.

In addition to prey species, domestic cat populations are likely to have a variety of direct and indirect impacts on native predators. Negative interactions are suggested by research showing that bobcats and domestic cats have limited coexistence in a variety of land cover types. For example, in riparian oak woodlands adjacent to vineyards in Northern California, Hilty and Merenlender (2006) have shown that sites where bobcats were detected did not have domestic cat populations. In Southern California, Crooks (2002) has shown an apparent lack of coexistence between native predators and domestic cats across a gradient of urban habitat

fragments, suggesting domestic cats and bobcats co-occur less frequently than would be expected by chance.

There is some evidence to support several possible mechanisms of negative interactions between domestic cats and bobcats. For example, resource competition due to overlaps in diet may be likely. In Mediterranean climates, bobcats exhibit low diet diversity (91-99%) lagomorphs and rodents) relative to other native carnivores (Fedriani et al. 2000), while domestic cats exhibit a strong preference for native species of small mammals (Hall et al. 2000). Research on predation has shown that domestic cats continue to exploit prey populations even when local abundances are low (Churcher & Lawton 1987). Interference competition, or intraguild predation of domestic cats by bobcats, is also possible, as bobcats are a solitary species and generally maintain exclusive home ranges (Nowell and Jackson 1996). In addition, predation by a second sympatric predator may contribute to the exclusion of domestic cats from bobcat territories. In particular, bobcats often coexist with coyotes (Fedriani et al. 2000), and coyotes are frequent predators of domestic cats (Crooks & Soulé 1999). Finally, domestic cat populations may also serve as sources of disease for carnivore populations, particularly wild felids. Coastal contamination of southern sea otter populations with toxoplasmosis has been attributed to landbased surface runoff (Miller et al. 2001), and Feline Leukemia (FeLV) and Feline Distemper (FPV) have been diagnosed in mountain lions, Feline Peritonitis (FIP) in mountain lions and lynx, and Feline Immunodeficiency Virus (FIV) in mountain lions and bobcats (Jessup et al. 1993; Roelke et al. 1993).

One way to alleviate the impacts of domestic cat presence on wildlife in the SDC is to encourage the public to keep their pet cats indoors. Cat owners may not be aware of the adverse effects the animals have on wildlife, so a campaign explaining the effects of predation and

disease transmission by cats might raise awareness and thus persuade owners to decrease their pet's access to the outdoors. Additionally, depending on the presence of feral cats in the area, another way to potentially mitigate the impacts cats have on wildlife is to manage the presence of feral cats in the area.

Dogs (*Canis lupus familiaris*) are allowed within protected areas in many countries worldwide, which can result in management concerns about dogs and their regulation to prevent wildlife disturbance and predation (Weston et al. 2014). In addition to being the most widespread canid (Silva-Rodríguez & Sieving 2012), dogs are adaptable, social carnivores. As carnivores, dogs have the potential to negatively impact a park ecosystem by disturbing, preying upon, and competing with wildlife. By interacting in these ways, dogs may influence the composition of wildlife populations, which is of particular concern when parks are home to sensitive, endemic, or endangered species. For example, in a study investigating the effects of dogs on native mammalian carnivores in parks, researchers found the relative abundance of native coyotes (*Canis latrans*) and bobcats (*Lynx rufus*) were four times greater in sites with no public access (Reed & Merenlender 2011).

Whether dogs should even be allowed in urban parks is controversial. Dog owners advocate for dog-friendly access with fewer restrictions (Slater et al. 2008; Kubinyi et al. 2009), whereas non-dog owners prefer to limit dog access with increased regulations (Instone and Mee 2011). Regulations that could address concerns about disturbance, human safety and dog waste include a combination of dog management and visitor compliance such as temporal and spatial restrictions of dogs, leashing regulations, and codes of conduct. However, compliance with park regulations by visitors with dogs is low. In a review of 22 published studies investigating compliance with "on leash" regulations in parks, most studies reported low compliance (63.3%),

and 36.4% reported medium compliance. None of the studies included in the review reported high compliance (Weston et al. 2014).

Recreation impacts

Outdoor recreation and ecotourism are increasingly popular, and access to parks and green spaces has many positive effects for humans (Nilsson 2006; O'Brien & Snowdon 2007). In contrast to the commonly-held assumption that non-motorized forms of habitat use for recreation, like hiking, biking, and horseback riding, are compatible with biodiversity conservation, there is a growing body of evidence showing negative impacts on wildlife (Losos et al. 1995; Reed & Merenlender 2008; Steven & Castley 2013). In a review of 218 articles, Larson et al. (in prep.) quantified the effects of recreation on wildlife as reported across a global distribution, without restrictions on taxonomic groups influenced or type of recreation examined. Over 93% of the reviewed articles documented at least one effect of recreation on wildlife, with negative effects most frequently reported (59.2% of studies); the extent of the effect varied with recreation activity and animal(s) studied. One surprising finding was that non-motorized activities, like hiking, had more evidence for a recreation effect than motorized activities. Despite this evidence of negative impacts on wildlife, Larson et al. found that 35% of the reviewed articles did not provide accompanying management recommendations. The review also highlighted gaps in our understanding about recreation impacts, such as a need for additional research that include animals of conservation concern and community-level investigations (Larson et al. in prep).

Recreation activity on trails and roads may lead to indirect habitat loss for wildlife as animals avoid areas frequented by humans (Hebblewhite & Merrill 2008). Further, the impacts of human activity are complicated, with differing responses by animals across taxa and trophic

level. While many animals universally avoid habitat on or directly adjacent to trails and roads, land use by predator and prey species can differ with increasing distance from trails. For example, at low levels of human activity (i.e., less than two people/hour) wolves avoid areas of intermediate distance (50 -400m) from trails, whereas elk, their prey species, use these areas as predation refugia. When recreation intensity increased to two people/hour both species avoided trails and all habitat up to 400m from trails (Rogala et al. 2011).

Human-wildlife interactions can also cause physiological stress in animals, which may interfere with survival and reproduction. In a review of the environmental effects of wildlife viewing, hiking, and cycling on birds, researchers found overwhelming evidence of negative effects of these activities (Steven et al. 2011). Of the 69 papers included in the review, 88% found negative impacts such as changes in behavior (90%; 37 out of 41 papers) and reproductive success (85%; 28 out of 33 papers) in birds exposed to these non-motorized recreation activities (Steven et al. 2011). Similar results have been shown for terrestrial animals. In a study of 28 protected areas in Marin, Sonoma, and Napa Counties in northern California (122° 12' to 122° 51' W, 38° 0' to 38° 37' N), Reed and Merenlender (2008) showed that the presence of quiet, non-motorized recreation led to a five-fold decline in native carnivore density, and caused a shift in community composition from native to nonnative species.

Research has shown that even low levels of human-wildlife interaction can produce measurable levels of physiological stress. The physiological stress experienced by animals exposed to recreation and tourism can been measured by analyzing concentrations of fecal glucocorticoids and their metabolites. For example, wildcats in zones of restricted human use in a natural park showed increased levels of stress – as measured by cortisol levels – with tourism intensity in a natural park (Piñeiro et al. 2012).

While understanding the potentially negative impacts of recreation, it is also critical to focus on the human values, attitudes, and behaviors motivating recreational users. We need to find a balance between continuing the public good of access to parks and forests, while also mitigating ecological disturbance caused by recreation for management to be effective and socially acceptable (Decker et al. 2009). For example, management that promotes responsible and respectful recreation may be more successful than efforts to curb behavior that threatens the ecological integrity of the habitat (Marzano & Dandy 2012).

One way to manage wildlife habitat is through the use of spatial restrictions on human activities in the form of fencing, designating trail-free areas, implementing leash laws, and increased management to ensure people use and stay on established trails.

Spatial restrictions that limit human access to wildlife habitat provide animals with a refuge from human recreational activities. Barriers restricting human-wildlife contact can be simple, affordable means to provide animals with a refuge from human recreational activities. For example, human contact with birds can negatively impact bird survival by causing birds to avoid feeding areas (Gill et al. 1996), provide inferior parental care (Verhulst et al. 2001), and demonstrate increased stress in the form of elevated heart rate (Culik et al., 1990). Protective barriers that restrict human access to bird habitat provides areas of refuge for birds, allowing them to behave as they would in an undisturbed environment (Ikuta & Blumstein 2003).

In addition to physically separating wildlife from human disturbance, temporal restrictions on recreation may also protect animals. Habitat that may be open to public use during most of the year may be temporarily closed during seasonal migrations or a sensitive breeding period. For example, access to numerous beaches along the Pacific Coast in California is restricted to protect elephant seals during their breeding season. If the beach is a State or

National Park or Reserve, a park ranger is present during restricted access season to provide public outreach by explaining that the restricted beach access is to protect the elephant seals and describing the ecology and life history of the animal. Temporal restrictions on park use and visitor number during animals' sensitive gestation period have also been recommended for terrestrial animals like the wildcat in Spain (Pineiro et al. 2012).

Given the importance of the SDC as a crucial component of the Sonoma Valley Wildlife Corridor, management that mitigates the negative impacts of roads, nocturnal lights, domestic cats and dogs, and human recreation is essential to preserve the integrity of this habitat as a wildlife corridor.

Conclusion

Our key findings are as follows:

Landscape permeability

- Much of the northern portion of the SDC has high estimates for landscape permeability, and hence is expected to allow for free passage of wildlife if left undisturbed.
 Specifically, 32% of the landscape had the highest permeability values, indicating there is land of relatively high permeability within the SDC property, and such habitat is not rare.
- Areas where permeability is likely compromised by development span a gradient of low to intermediate permeability values.

Climate benefit analysis

• Three ways the resilience of a reserve network to climate change may be improved are by prioritizing corridors that: 1) provide access to cooler climates, 2) maintain continuous

habitat across a diversity of climate types, and 3) maintain access to areas with slower rates of change.

- The SDC overlaps with a corridor (c632) identified as part of the Mayacamas
 connectivity plan that connects Sonoma Mountain with the Southern Mayacamas
 Mountains (patches p534 and p474, respectively). Protecting the SDC would widen the
 proposed corridor c632, offering additional connectivity in this bottleneck between the
 habitat patches. Climate analysis is reported for this corridor from existing analysis done
 by <u>The Terrestrial Biodiversity and Climate Change Collaborative (TBC3)</u>, a group of
 university, NGO, and governmental researchers (Merenlender et al. in prep).
- Summer temperatures (JJA): Access to cooler habitat during warm summer months is important for mobile animals in the immediate term and dispersing plants and animals in the long term, particularly in light of changing climates. Historically, this corridor provided between 1.52 1.59 °C cooling during warm summer months in the years 1951 2010. This corridor is estimated to provide access to cooler coastal areas that are between 1.02 1.06 °C cooler during warm summer months in the future long term predictions for 2070 2099.
- Winter temperatures (DJF): Access to warmer habitat during cool winter months is important for mobile animals in the immediate term and dispersing plants and animals in the long term, particularly in light of changing climates. Historically, this corridor provided between 0.1 0.19 °C cooling during winter months in the years 1951- 2010. This corridor provides access to cooler higher elevation areas that are estimated to be about 0.12 °C cooler for winter minimum temperatures in the future long-term predictions for 2070 2099.

- *Climatic diversity*: Climate water deficit (CWD) is an integrated metric of climatic variables that influence vegetation. The diversity of CWD levels is correlated with biological processes as well as the distribution of plants and animals. Historically, the climate diversity of both upland habitat patches connected by the corridor revealed CWD diversity levels to be 41.24 42.28 higher if the two boarding habitat patches are connected than if they remain isolated. Long-term predictions for 2070 2099 show maintenance of this diversity advantage (42.75 44.47) and increasing CWD over time. These are large differences in available CWD diversity as compared with other similarly sized linkages throughout the North Bay.
- Speed of climate change: Increased access to a wider range of elevations also slows the effective rate of climate change within the two patches connected by the Sonoma Valley Wildlife corridor, and in this case the Sonoma Valley Wildlife Corridor would provide a reduction in the velocity of climate change of 0.11 km/year between historical averages and 2070 2099 averages. Areas with more stable climates offer a greater chance for local adaptation.

Built environment analysis

- The majority of the buildings (n=172) are clustered in the center of the SDC footprint, along the southern border of the property impacting an estimated 7954.52 square feet or 182.82 acres. There is a smaller cluster of (n=41) buildings in the eastern flank of the property impacting an estimated 699.65 square feet or 16.18 acres.
- High building and road density in the center of the SDC effectively creates a bottleneck for wildlife movement along the northern border of the property that is at most 689 feet wide and 2560 feet long.

The developed areas identified here will support more birds adapted to urban conditions and a greater degree of exotic plants that have less ecological value to native insects and vertebrates. Further, the distribution and abundance of mammal species would change such that many carnivores, including bobcats (*Lynx rufus*), coyotes (*Canis latrans*), and pumas (*Puma concolor*), could decline or disappear if development levels intensify. Removing isolated buildings and any not required for future use is highly recommended to enhance wildlife movement and the overall ecological integrity of the SDC property.

Managing for connectivity

- *Roads and traffic:* Roads and traffic have an overwhelming negative impact on animal populations. Traffic calming is one way to improve road safety and mitigate the negative impacts of roads.
- *Nighttime lights:* Artificial nighttime lighting has been shown to alter individual animal and bird behavior and diminishes reproductive success and survivorship. To mitigate the negative impacts of artificial night time lighting at the SDC, the number of active lighting fixtures and the intensity of their bulbs could be decreased.
- *Wildlife-friendly fencing:* Improper fence design can result in animal injury or death as a result of collision or entanglement across a wide variety of animals and birds. Guidelines exist for wildlife friendly fencing that increase fence visibility and prevent animal ensnarement and mortality.
- *Domestic cat presence:* Free-roaming and feral domestic cats are the most significant exotic predators worldwide. Field data on free-ranging domestic cats reveal that some individuals can kill over 1000 wild animals per year, spread disease and are associated with development and high human activity rates. One way to alleviate the impacts of

domestic cat presence on wildlife in the SDC is to encourage the public to keep their pet cats indoors and manage the presence of feral cats in the area.

- Domestic dog presence: Human-accompanied dogs are allowed within protected areas in many countries worldwide, which has resulted in management concerns about their ecological impact. Proposed regulations that could address concerns about wildlife disturbance, human safety, and dog waste in parks include temporal and spatial restrictions of dogs, and leash requirements.
- *Recreation impacts:* Human recreation activities have been shown to have negative impacts on wildlife, including indirect habitat loss for wildlife as animals avoid areas frequented by humans, as well as physiological stress in animals that may interfere with survival and reproduction. In order for management to be effective and socially acceptable, it is critical to develop a recreation plan that provides the benefits of access to public lands while also mitigating ecological disturbance caused by recreation within this crucial corridor. Limiting human access to wildlife habitat in the form of fencing, designated trail-free areas, leash laws, and increased management to ensure people stay on trails provides animals with a refuge from human recreational activities. Additionally closing trails to public use during seasonal migrations, sensitive breeding periods or high-use times may protect animals.

Maps and illustrations

1. Location map

This map of the SDC property includes buildings in red with a gray outline mapped using LiDAR imagery taken in 2014 by Sonoma County. Streets and contouring are available through ESRI mapping tools and open source information data sets.



2. Built environment impact envelope map

This built environment impact map of the SDC property identifies buildings and roads with an impact buffer of 30m surrounding each feature.



3. Landscape permeability map

Landscape permeability map overlaid on the Sonoma Developmental Center footprint.



4. Climate space map

Map showing the geographic configuration between patches p534 and p474, corridor c632, and the Sonoma Developmental Corridor.



References

- Adriaensen, F., J. P. Chardon, G. De Blust, E. Swinnen, S. Villalba, H. Bulinck, and E. Matthysen. 2003. The application of "least-cost" modelling as a functional landscape model. Landscape and Urban Planning 64:233–247.
- Ahern, J. 1995. Greenways as a planning strategy. Landscape and Urban Planning 33:131-155.
- Alonso, R. S., L. M. Lyren, E. E. Boydston, C. D. Haas, and K. R. Crooks. 2014. Evaluation of road expansion and connectivity mitigation for wildlife in southern California. The Southwestern Naturalist 59:181–187.
- American Bird Conservancy (ABC). 2002. Domestic cat predation on birds and other wildlife. American Bird Conservancy, Washington, D.C.
- Arango-Velez, N., and G.H. Kattan. 1997. Effects of forest fragmentation on experimental nest predation in Andean cloud forest. Biological Conservation 81:137-143.
- Baker, B. J., and J. M. L. Richardson. 2006. The effect of artificial light on male breeding-season behaviour in green frogs, Rana clamitans melanota. Canadian Journal of Zoology 84:1528– 1532.
- Beier, P. 1993. Determining minimum habitat areas and habitat corridors for cougars. Conservation Biology 7:94-108.
- Beier, P. 2005. Chapter 2 Effects of artificial night lighting on terrestrial mammals:1–24.
- Beier, P., and B. Brost. 2010. Use of Land Facets to Plan for Climate Change: Conserving the Arenas, Not the Actors. Conservation Biology 24:701–710.
- Beier, P., D. Choate, and R. Barrett. 1995. Movement patterns of mountain lions during different behaviors. Journal of Mammalogy:1056–1070.
- Bender, D. J., L. Tischendorf, and L. Fahrig. 2003. Using patch isolation metrics to predict animal movement in binary landscapes. Landscape Ecology 18:17–39.
- Bennett, A. 1999. Linkages in the landscape: the role of corridors and connectivity in wildlife conservation.
- Bissonnette, T., and A. Csech. 1938. Modified sexual photoperiodicity in cotton tail rabbits:364–367.
- Boone, R. B., and N. T. Hobbs. 2004. Lines around fragments: effects of fencing on large herbivores. African Journal of Range & Forage Science 21:147–158.

- Bunn, D., A. Mummert, M. Hoshovsky, K. Giliardi, and S. Shanks. 2007. California wildlife: conservation challenges. California Department of Fish and Game. 597pp. http://www.dfg.ca.gov/SWAP/2005/docs/SWAP-2005.pdf
- Burbidge, A. A., and B. F. J. Manly. 2002. Mammal extinctions on Australian islands: Causes and conservation implications. Journal of Biogeography 29:465–473.
- Burrows, M. T. et al. 2014. Geographical limits to species-range shifts are suggested by climate velocity. Nature 507:492–5.
- Butchart, S. et al. 2010. Global Biodiversity: Indicators of Recent Declines. Science (New York, NY) 328:1164–1168.
- Calabrese, J. M., and W. F. Fagan. 2004. A comparison-shopper's guide to connectivity metrics. Frontiers in Ecology and the Environment 2:529–536.
- California Climate Adaptation Strategy (CCAS). 2009. California Natural Resources Agency. 200pp. http://www.climatechange.ca.gov/adaptation/documents/Statewide_Adaptation_Strategy_____Chapter_5_-_Biodiversity_and_Habitat.pdf
- Churcher, P., and J. Lawton. 1987. Predation by domestic cats in an English village. J Zool 212:439–455.
- Crooks, K.R., and M.E. Soulé. 1999. Mesopredator release and avifaunal extinctions in a fragmented system. Nature 400:563-566.
- Crooks, K., C. Burdett, D. Theobald, C. Rondinini, and L. Boitani. 2011. Global patterns of fragmentation and connectivity of mammalian carnivore habitat. Philosophical Transactions of the Royal Society B: Biological Sciences 366:2642–2651.
- Coleman, J. S. and S. A. Temple. 1995. How many birds do cats kill? Wildlife Control Technology 2: 44.
- Crooks, K. R. 2002. Relative sensitivities of mammalian carnivores to habitat fragmentation. Conservation Biology 16:488–502.
- Crooks, K. R., and M. E. Soulé. 1999. Mesopredator release and avifaunal extinctions in a fragmented system. Nature 400:563–566.
- Culik, B., D. Adelung, and A. J. Woakes. 1990. The Effect of Disturbance on the Heart Rate and Behaviour of Adélie Penguins (*Pygoscelis adeliae*) During the Breeding Season. In: Kerry, K. R. and G. Hempel (eds) Antarctic Ecosystems: Ecological change and conservation. pp177-182.

- Daly, C., M. Halbleib, J. I. Smith, W. P. Gibson, M. K. Doggett, G. H. Taylor, J. Curtis, and P. P. Pasteris. 2008. Physiographically sensitive mapping of climatological temperature and precipitation across the conterminous United States. International Journal of Climatology 28:2031–2064.
- Decker, D. J., W. F. Siemer, K. Leong, S. J. Riley, B. A. Rudolph, and L. H. Carpenter. 2009. Conclusion: What is wildlife management? In: Manfredo M. J., J. J. Vaske, P. J. Brown, D. J. and E. A. Duke (eds) Wildlife and society: the science of human dimensions. Island Press, Washington DC, pp 315-327.
- Dijak, W.D., and F.R. Thompson. 2000. Landscape and edge effects on the distribution of mammalian predators in Missouri. Journal of Wildlife Management 64: 209-216.
- Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. Annual Review of Ecology, Evolution, and Systematics 34:487–515.
- Fahrig, L., and T. Rytwinski. 2009. Effects of roads on animal abundance: an empirical review and synthesis. Ecology & Society 14.
- Farmland Mapping & Monitoring Program, 2008. http://redirect.conservation.ca.gov/DLRP/fmmp/county_info_results.asp. Accessed: May 2011.
- Fedriani, J. M., T. K. Fuller, R. M. Sauvajot, and E. C. York. 2000. Competition and intraguild predation among three sympatric carnivores.
- Flint, L. E., and A. L. Flint. 2012. Downscaling future climate scenarios to fine scales for hydrologic and ecological modeling and analysis. Ecological processes 1:1–15.
- Forman, R. T. T. 2000. Estimate of the area affected ecologically by the road system in the United States. Conservation Biology 14:31–35.
- Forman, R. T. T., and R. D. Deblinger. 1998. The ecological road-effect zone for transportation planning and Massachusetts highway example.
- Fu, W., S. Liu, S. D. Degloria, S. Dong, and R. Beazley. 2010. Characterizing the fragmentation

 a effect of road networks on l
 Xishuangbanna, Southwest China. Landscape and Urban Planning 95:122–129.
- Garrison, B. A. 2000. Draft Report: A Strategy for Conserving Oak Woodlands in Vineyard Landscapes. California Dept. of Fish and Game. 36 p.
- Gill, J. A., W. J. Sutherland, and A. R. Watkinson. 1996. A method to quantify the effects of human disturbance on animal populations. Journal of Applied Ecology 33:786–792.

- Girvetz, E. H., J. H. Thorne, A. M. Berry, and J. A. G. Jaeger. 2008. Integration of landscape fragmentation analysis into regional planning: A statewide multi-scale case study from California, USA. Landscape and Urban Planning 86:205–218.
- Gloyne, C.C. & Clevenger, A.P. 2001. Cougar *Puma concolor* use of wild-life crossing structures on the Trans-Canada highway in Banff National Park, Alberta. Wildlife Biology 7: 117-124.
- Goad, E. H., Pejchar, L., Reed, S. E., & Knight, R. L. 2014. Habitat use by mammals varies along an exurban development gradient in northern Colorado. Biological Conservation, 176:72-182.
- Gray, M., C. C. Wilmers, S. E. Reed, and A. M. Merenlender. Accepted. Landscape featurebased permeability models relate to puma occurrence. Landscape and Urban Planning.
- Hall, L. S., M. A. Kasparian, D. Van Vuren, and D. A. Kelt. 2000. Spatial organization and habitat use of feral cats (Felis catus L.) in Mediterranean California.
- Hansen, A.J., R. Rasker, B. Maxwell, J.J. Rotella, J.D. Johnson, A. Wright Parmenter, U. Langner, W.B. Cohen, R.L. Lawrence, and M.P.V. Kraska. 2002. Ecological causes and consequences of demographic change in the new west. BioScience 52:151-162.
- Harris, L.D., T. Hoctor, D. Maehr, and J. Sanderson. 1996. The role of networks and corridors in enhancing the value and protection of parks and equivalent areas. Pages 173-197 in R.G. Wright, ed., National Parks and Protected Areas: Their Role in Environmental Protection. Cambridge, Massachusetts: Blackwell Scientific Publications.
- Hawkins, C. C. 1998. Impact of a subsidized exotic predator on native biota: effect of house cats (Felis catus) on California birds and rodents. Ph. D. dissertation. Texas A and M University, College Station, TX.
- Heller, N., and E. Zavaleta. 2009. Biodiversity management in the face of climate change: A review of 22 years of recommendations. Biological Conservation 142:14–32.
- Henle, K., Lindenmayer, D.B.; Margules, C.R., Saunders, D.A., and C. Wissel. 2004. Species survival in fragmented landscapes: where are we now? Biodiversity and Conservation 13:1-8.
- Hilty, J. A., C. C. Chester, and M. S. Cross. 2012. Climate and Conservation: Landscape and Seascape Science, Planning, and Action. Page 392. Island Press.
- Hilty, J. A., W. Z. L. Jr., and A. Merenlender. 2006. Corridor Ecology: The Science and Practice of Linking Landscapes for Biodiversity Conservation. Page 344. Island Press.
- Hölker, F., C. Wolter, E. K. Perkin, and K. Tockner. 2010. Light pollution as a biodiversity threat. Trends in ecology & evolution 25:681–2.

- Huber, P. R., F. Shilling, J. H. Thorne, and S. E. Greco. 2012. Municipal and regional connectivity planning. Landscape and Urban Planning 105: 15-26.
- Ikuta, L. A., and D. T. Blumstein. 2003. Do fences protect birds from human disturbance? Biological Conservation 112:447–452.
- IUCN. 2013. IUCN Red List of Threatened Species. Available from www.iucnredlist.org.
- Jaarsma, C. F., F. Van Langevelde, and R. Beunen. 2013. Landscape Ecology and Rural Roads : Traffic Calming for improving both landscape and wildlife ? 32:352–360.
- Jackson Hole Wildlife Foundation (JHWF). 2013. Wildlife friendly fencing for Teton County: A resource notebook for the Jackson Hole Community.
- Jessup, D. A., K. C. Pettan, L. J. Lowenstine, and N. C. Pedersen. 1993. Feline leukemia-virus infection and renal spirochetosis in a free-ranging cougar (*Felis concolor*). Journal of Zoo and Wildlife Medicine 24:73–79.
- Joly, P., C. Morand, and A. Cohas. 2003. Habitat fragmentation and amphibian conservation: building a tool for assessing landscape matrix connectivity. Comptes Rendus Biologies 326:132–139.
- Kaiser, J. 2001. Bold corridor project confronts political reality. Science (New York, NY) 293:2196–2199.
- Klar, N., M. Herrmann, M. Henning-Hahn, B. Pott-Dörfer, H. Hofer, and S. Kramer-Schadt Stephanie. 2012. Between ecological theory and planning practice: (Re-) Connecting forest patches for the wildcat in Lower Saxony, Germany. Landscape and Urban Planning 105:376–384.
- Kramer, K. M., and E. C. Birney. 2001. Effect of light intensity on activity patterns of Patagonian leaf-eared mice, *Phyllotis xanthopygus*. Journal of Mammalogy 82:535–544.
- Kubinyi, E., P. Pongrácz, and Á. Miklósi. 2009. Dog as a model for studying conspecific and heterospecific social learning.
- LaRue, M., and C. Nielsen. 2008. Modelling potential dispersal corridors for cougars in midwestern North America using least-cost path methods. Ecological Modelling 212:372– 381. Elsevier B.V., Cooperative Wildlife Research Laboratory, Southern Illinois University Carbondale, Carbondale, IL, USA.
- Larson, C. L., S. E. Reed, A. M. Merenlender, and K. R. Crooks. In preparation. Effects of recreation on animals revealed as widespread through a global systematic review.
- Laurance, W.F. 1995. Landscape Approaches in Mammalian Ecology and Conservation. U of Minnesota Press.

- Laurance, W.F. 1997. Hyper-disturbed parks: edge effects and the ecology of isolated rainforest reserves in tropical Australia. Pages 71-83 in W.F. Laurence and R.O.J. Bierregaard, eds., Tropical Forest Remnant. Chicago, Illinois: University of Chicago Press.
- Laurance, W.F., T.E. Lovejoy, H.L. Vasconcelos, E.M. Bruna, R.K. Didham, P.C. Stouffer, C. Gascon, R.O. Bierregaard, S.G. Laurance, and E. Sampaio. 2002. Ecosystem Decay of Amazonian Forest Fragments: a 22-year investigation. Conservation Biology 16:605-618.
- Lawler, J. J., a S. Ruesch, J. D. Olden, and B. H. McRae. 2013. Projected climate-driven faunal movement routes. Ecology letters 16:1014–22.
- Loarie, S. R., P. B. Duffy, H. Hamilton, G. P. Asner, C. B. Field, and D. D. Ackerly. 2009. The velocity of climate change. Nature 462:1052–1055.
- Longcore, T. 2010. Sensory ecology: night lights alter reproductive behavior of blue tits. Current biology : CB 20:R893–5.
- Longcore, T., and C. Rich. 2004. Ecological Light Pollution. Frontiers in Ecology and the Environment 2:191.
- Marzano, M., and N. Dandy. 2012. Recreationist behaviour in forests and the disturbance of wildlife. Biodiversity and Conservation 21:2967–2986.
- Mcintyre, S., and R. Hobbs. 1999. A framework for conceptualizing human effects on landscapes and its relevance to management and research models. Conservation Biology 13:1282–1292.
- McRae, B., B. Dickson, T. Keitt, and V. Shah. 2008. Using circuit theory to model connectivity in ecology, evolution and conservation. Ecology 89:2712–2724.
- Merenlender, A. M., K. L. Heise, and C. Brooks. 1998. Effects of subdividing private property on biodiversity in California's north coast oak woodlands. Transactions of the Western Section of the Wildlife Society 34:9-20.
- Merenlender, A. M., D. Ackerly, S. Feirer, M. Gray, and J. Kreitler. In preparation. Assessing local habitat corridors for climate change resilience and protected area planning.

Merenlender, A. M. 2011a. Santa Cruz Mountains Connectivity Report:30.

- Merenlender, A. M. 2011b. Santa Cruz County Threats and Costs Analysis Report:12.
- Merenlender, A. M., S. Reed, and K. Heise. 2009. Exurban development influences woodland bird composition. Landscape and Urban Planning 92:255–263.

- Merenlenderr, A. M., Brooks, C., Shabazian, D., Gao, S., and R. Johnston. 2005. Forecasting exurban development to evaluate the influence of land-use policies on wildland and farmland conservation. Journal of Conservation Planning 1(1):64-88.
- Mesquita R.C.G., Delamonica P. and Laurance W.F. 1999. Effect of surrounding vegetation on edgerelated tree mortality in Amazonian forest fragments. Biological Conservation 91:129–134.
- Metzger, J.-P., and H. Décamps. 1997. The structural connectivity threshold: An hypothesis in conservation biology at the landscape scale.
- Miller, M. A., I. A. Gardner, C. Kreuder, D. M. Paradies, K. R. Worcester, D. A. Jessup, E. Dodd, M. D. Harris, J. A. Ames, A. E. Packham and P. A. Conrad. 2002. Coastal freshwater runoff is a risk factor for Toxoplasma gondii infection of southern sea otters (Enhydra lutris nereis). International Journal for Parasitology 32: 997-1006.
- Nassar, R., and J. Mosier. 1991. Projections of pet populations from census demographic data. Journal of the American Veterinary Medical Association 198:1157–1159.
- Nilsson, C., C. A. Reidy, M. Dynesius, and C. Revenga. 2005. Large River Systems Fragmentation and Flow Regulation of the World's. Science 308:405–409.
- Nilsson, K. 2006. Forests, trees and human health and wellbeing. Urban Forestry and Urban Greening 5:109.
- Noss, R. 2001. Beyond Kyoto: forest management in a time of rapid climate change. Conservation Biology 15:578–590.
- Nowell, K. and P. Jackson. 1996. Wild Cats. IUCN, Gland, Switzerland.
- Nuñez, T. a, J. J. Lawler, B. H. McRae, D. J. Pierce, M. B. Krosby, D. M. Kavanagh, P. H. Singleton, and J. J. Tewksbury. 2013. Connectivity planning to address climate change. Conservation biology : the journal of the Society for Conservation Biology 27:407–16.
- O'Brien, L., and H. Snowdon. 2007. Health and well-being in woodlands: a case study of the Chopwell Wood health project.
- Odell, E. A., and R. L. Knight. 2001. Songbird and Medium-Sized Mammal Communities Associated with Exurban Development in Pitkin County, Colorado. Conservation Biology 15:1143–1150.
- Paige, C. 2008. A Landowner's Guide to Wildlife Friendly Fences. Landowner/Wildlife Resources Program. Montana Fish, Wildlife, and Parks, Helena, MT. 44pp.
- Panetta, F. D., and A. J. M. 1991. Hopkins. Weeds in corridors: invasion and management. Nature conservation 2:341-351.

- Piñeiro, A., I. Bárja, G. Silvn, and J. C. Illera. 2012. Effects of tourist pressure and reproduction on physiological stress response in wildcats: Management implications for species conservation. Wildlife Research 39:532–539.
- Rao, R. C. 1984. Use of diversity and distance measures in the analysis of qualitative data. Pages 49–67 Multivariate Statistical Methods in Physical Anthropology.
- Ray, N., A. Lehmann, and P. Joly. 2002. Modeling spatial distribution of amphibian populations: a GIS approach based on habitat matrix permeability. Biodiversity and Conservation 11:2143–2165.
- Reed, S. E. 2007. Dissertation:1–134.
- Reed, S. E., and A. M. Merenlender. 2011. Effects of Management of Domestic Dogs and Recreation on Carnivores in Protected Areas in Northern California. Conservation biology :
- Reed, S., and A. Merenlender. 2008. Quiet, Nonconsumptive Recreation Reduces Protected Area Effectiveness. Conservation Letters:1–9.
- Reijnen, R., R. Foppen, C. T. Braak, and J. Thissen. 1995. The effects of car traffic on breeding bird populations in woodland. III. Reduction of density in relation to the proximity of main roads.
- Reijnen, R., R. Foppen, and H. Meeuwsen. 1996. The effects of traffic on the density of breeding birds in Dutch agricultural grasslands. Biological conservation 75:255–260.
- Research and Innovative Technology Administration, Bureau of Transportation Statistics. http://www.rita.dot.gov/bts/data_and_statistics/databases. Accessed: May 2011.
- Ribeiro, M., R. Metzger, A. Martensen, F. Ponzoni, and M. Hirota. 2009. The Brazilian Atlantic Forest: How much is left, and how is the remaining forest distributed? Implications for conservation. Biological Conservation 142:1141–1153.
- Ricketts, T. 2009. The matrix matters: Effective isolation in fragmented landscapes. The American Society of Naturalists 158:87–99.
- Roelke, M. E., D. J. Forrester, E. R. Jacobson, G. V Kollias, F. W. Scott, M. C. Barr, J. F. Evermann, and E. C. Pirtle. 1993. Seroprevalence of infectious disease agents in freeranging Florida panthers (Felis concolor coryi). Journal of wildlife diseases 29:36–49.
- Rogala, J. K., M. Hebblewhite, J. Whittington, C. A. White, J. Coleshill, and M. Musiani. 2011. Human activity differentially redistributes large mammals in the Canadian Rockies national parks. Ecology and Society 16:17.

- Safner, T., C. Miaud, O. Gaggiotti, S. Decout, D. Rioux, S. Zundel, and S. Manel. 2011. Combining demography and genetic analysis to assess the population structure of an amphibian in a human-dominated landscape. Conservation Genetics 12:161–173.
- Sharma, V. K., M. K. Chandrashekaran, and P. Nongkynrih. 1997. Daylight and artificial light phase response curves for the circadian rhythm in locomotor activity of the field mouse *Mus booduga*. Biological Rhythm Research 28(Supplement):39–49.
- Silva-Rodríguez, E. A., and K. E. Sieving. 2012. Domestic dogs shape the landscape-scale distribution of a threatened forest ungulate. Biological Conservation 150:103–110.
- Sizer, N., and E.V.J. Tanner. 1999. Responses of woody plant seedlings to edge formation in a lowland tropical rainforest, Amazonia. Biological Conservation 91:135-142.
- Slater, M. R., A. Di Nardo, O. Pediconi, P. D. Villa, L. Candeloro, B. Alessandrini, and S. Del Papa. 2008. Cat and dog ownership and management patterns in central Italy. Preventive Veterinary Medicine 85:267–294.
- Sonoma County Agrigultural Preservation and Open Space District (SCAPOSD). 2014. A landowners guide: fencing and other solutions for protecting and promoting natural resources and wildlife.
- Stefan, A. 1999. Invasion of matrix species in small habitat patches. Conservation Ecology 3:1-14.
- Stephenson, N. L. 1998. Actual evapotranspiration and deficit: biologically meaningful correlates of vegetation distribution across spatial scales. Journal of Biogeography 25:855– 870.
- Steven, R., C. Pickering, and J. Guy Castley. 2011. A review of the impacts of nature based recreation on birds. Journal of Environmental Management 92:2287–2294.
- Tannier, C., J. C. Foltête, and X. Girardet. 2012. Assessing the capacity of different urban forms to preserve the connectivity of ecological habitats. Landscape and Urban Planning 105:128–139.
- Taylor, P. D., L. Fahrig, K. Henein, and G. Merriam. 1993. Connectivity is a vital element of landscape structure. Oikos:571–573.
- Theobald, D. M., S. E. Reed, K. Fields, and M. Soulé. 2012. Connecting natural landscapes using a landscape permeability model to prioritize conservation activities in the United States. Conservation Letters 5:123–133.
- Theobald, D. M. 2005. Landscape patterns of exurban growth in the USA from 1980 to 2020. Ecology and Society 10:32.

- Tilman, D., J. Fargione, B. Wolff, C. D'Antonio, a Dobson, R. Howarth, D. Schindler, W. H. Schlesinger, D. Simberloff, and D. Swackhamer. 2001. Forecasting agriculturally driven global environmental change. Science (New York, N.Y.) 292:281–284.
- Underwood, J. G., J. Francis, and L. R. Gerber. 2011. Incorporating biodiversity conservation and recreational wildlife values into smart growth land use planning. Landscape and Urban Planning 100:136–143.
- Verhulst, S., K. Oosterbeek, and B. J. Ens. 2001. Experimental evidence for effects of human disturbance on foraging and parental care in oystercatchers. Biological Conservation 101:375–380.
- Walker, R., and L. Craighead. 1998. "Corridors: key to wildlife from Yellowstone to Yukon." A sense of place: an atlas of issues attitudes, and resources in the Yellowstone to Yukon ecoregion. Yellowstone to Yukon Conservation Initiative, Canmore, Alberta. p113-121.
- Western Governors' Association (WGA). 2008. Wildlife Corridors Initiative. National Park Service, Department of the Interior. 142pp. http://www.nature.nps.gov/biology/migratoryspecies/documents/WGAWildlifeCorridorsIni tiative.pdf
- Weston, M. A., J. A. Fitzsimons, G. Wescott, K. K. Miller, K. B. Ekanayake, and T. Schneider. 2014. Bark in the park: A review of domestic dogs in parks. Environmental Management 54:373–382.
- Wiens, J. A. 1996. Wildlife in patchy environments: metapopulations, mosaics, and management. Pages 53-84 in D.R. McCullough, ed., Metapopulations and Wildlife Conservation. Covelo, California: Island Press.