Dissertation Proposal **Chris Hamilton**

Overview

Humans have directly influenced over 83% of the earth's land surface through land transformation activities that are one of the main threats to biological diversity (Sanderson et al. 2002). Land cover crossed the threshold of "mostly wild" to "mostly anthropogenic" within the last hundred years with less than 25% remaining in a wild state and over 39% in use for agriculture and human settlement (Ellis et al. 2010). These activities followed a similar pattern within the United States (Leu et al. 2008). The alteration of land cover and subsequent appropriation of the earth's resources is important to study because it entails impacts to climate, water quality and quantity, and biodiversity that affect management of our fish and wildlife resources (Theobald et al. 2009, Vitousek and Mooney 1997, Williams and Jackson 2007).

Conservation in the face of landscape change requires an understanding of key drivers of change, the threats the changes pose, and the potential tradeoffs associated with increased human presence (Wiens 2009, Heller and Zavaleta 2009). All of these impacts are exacerbated by human-induced climate change (Griffith et al. 2009). It is anticipated that we will see shifts in climate that include changes in the amount and timing of rainfall and the frequency of extreme events with likely impacts on ecological communities(Williams and Jackson 2007).While we are in the early stages of the projected changes in climate, there is already evidence of impacts to species, ecological communities, and ecosystems(Walther et al. 2002).

Adaptation to change necessitates evaluating impending changes to identify threats and appropriate responses (Smith et al. 2000). The incorporation of threat into conservation decisions is critical to maximizing conservation outcomes obtained from investing limited conservation funding (Merenlender et al. 2009b). Identification of future threats has been recognized as a priority research area for the U.S. Fish and Wildlife Service National Wildlife Refuge System (Griffith et al. 2009) and the potential future effects of land use and climate change on protected areas has been identified as one of the most important research areas needed to guide conservation policy (Fleishman et al. 2011). The combination of exploring potential scenarios and constructing predictive models has been touted as an important tool set to increase the value of ecological research for management application (Coreau et al. 2009). This approach can provide important information on the effects of alternate futures on biodiversity and other ecological resources (Gude et al. 2007, White et al. 1997).

Our current approach to conservation relies heavily on protected areas as refugia to safeguard biodiversity (Gaston et al. 2008). While protected areas alone are not sufficient for effective conservation, they are often the backbone of conservation (Margules and Pressey 2000). The global protected area network includes over 11% of the earth's land surface (Rodrigues et al. 2004). However, these protected areas, linked to their surroundings by ecological flows and processes, cannot be viewed in isolation (Hansen and DeFries 2007b).

Their effectiveness for conserving biodiversity is influenced by the surrounding landscape which is often in other intensive uses such as agriculture or human settlement (Griffith et al. 2009, Joppa et al. 2008, Wade and Theobald 2010). These surrounding land uses threaten, and may be limiting, the value and effectiveness of protected areas as a conservation tool (Joppa et al. 2008, Radeloff et al. 2010).

Large areas in the U.S are impacted by human activities due to land use patterns (Leu et al. 2008, Brown et al. 2005). The conversion of land for human use affects both ecological processes and biodiversity (Flather et al. 1998, Foley et al. 2005a). Intensive land use impacts biodiversity through both habitat loss and fragmentation (Fischer and Lindenmayer 2007, Fahrig 2003, Damschen et al. 2006). In addition, current and historic land use affect water quality (Locke et al. 2006, Allan 2004), community composition (Attum et al. 2008, Pidgeon et al. 2007), species range limits (Schulte et al. 2005), dispersal and movements (Damschen et al. 2006, Eigenbrod et al. 2008, Fahrig 2007), and invasion by non-native species (Gavier-Pizarro et al. 2010, Predick and Turner 2008).

Coupled human and natural systems, defined as those systems that incorporate interacting natural and human mechanisms and processes, are a product of human transformation of the earth (Liu et al. 2007a). Human choices and actions that affect landscape pattern have direct consequences for biodiversity (Pidgeon et al. 2007, Peterson et al. 2008). As such, effective conservation requires balancing human use and biodiversity at the landscape scale (Wiens 2009). Conservation and management opportunities that account for human needs while maintaining ecological function need to be identified (DeFries et al. 2007). In order to advance conservation, we need to understand future land use scenarios, to incorporate human-driven processes of land use change, to identify opportunities for, and potential vulnerabilities to, biodiversity conservation and landscape connectivity (Gude et al. 2007, Liu et al. 2007b).

Land use change around the world has followed a general pattern beginning with agricultural production in areas of high primary productivity, followed by development of industrial and population centers removed from agriculture as a second phase, and, finally, an information stage where land use is somewhat independent of primary productivity and industrial development (e.g. remote desk top) (Huston 2005). The landscape transitions through these stages from domination by natural ecosystems to intensive human use, and each stage has associated changes to ecosystem services such as fresh water, forest products, and food production (Foley et al. 2005b). This pattern of land use change is evident in the United States (Brown et al. 2005). The specific pattern within ecoregions has been modeled based on the linkages between land use history and ecological and economic processes. These models are designed to forecast land use change at an ecoregional level (Sohl et al. 2010) and can be used to assess the likely impact of different economic policies and scenarios on future land use patterns (Radeloff et al. in review). Forecasting land use change is essential to effective conservation planning and is most effective when conducted across multiple scales (Merenlender et al. 2009b, White et al. 1997, Foley et al. 2005b).

One particularly problematic aspect of land use intensification is housing growth and exurban development because it often occurs in a manner that results in high human-environment conflict (Radeloff et al. 2005a)(Radeloff et al. 2005b). Housing is a particularly persistent form of land use; once land use converts to housing it tends to stay in housing (Radeloff et al. 2010). Exacerbating negative impacts from exurban development is the fact that residential development has effects that extend beyond the footprint of the house. In addition to the houses themselves, residential development is normally tied to increased infrastructure developments that have their own associated impacts including: decreased native species diversity and abundance (Fahrig and Rytwinski 2009, Friesen et al. 1995, Lepczyk et al. 2008), increased rates of predation (Wilcove 1985), and interference with species movements (Fahrig 2007). These issues are compounded by the density and pattern of exurban housing growth, which often results in impacts to biodiversity that far exceed the actual footprint of the structures and lawn (Hansen et al. 2005, Gagne and Fahrig 2010a, Gagne and Fahrig 2010b). In addition, natural ecosystems, and especially protected areas, are attractive amenities that have been experiencing higher rates of housing growth than the rest of the United States (Wade and Theobald 2010, Radeloff et al. 2010). This means that our protected network is increasingly at risk of being broken apart into isolated units or smaller connected networks. This has serious implications for conservation because of disruption to ecological flows such as species movements and dispersal as well as the increasingly limited options it leaves for protected area management and adapting conservation to land use and climate change (Wiens 2009, Griffith et al. 2009, DeFries et al. 2007).

The question is how landscape connectivity can be retained in the face of land use change and housing growth. Connectivity is the relative spatial contagion of resource patches and how that contagion affects movements among and between resource patches (Crooks and Sanjayan 2006). When viewed most broadly, these movements include the flow of nutrients, energy, disturbance, and species. Connectivity is affected by the overall reduction in the amount of habitat and the breaking apart of habitat. While habitat fragmentation has variable effects on biodiversity, habitat loss negatively impacts biodiversity maintenance, with well-connected landscapes better maintaining biodiversity (Fahrig 2003). Connectivity for wildlife provides opportunities for normal home range movements, dispersal, gene flow, and adjustments to species range in response to process that occur at larger spatial and temporal scales (Crooks and Sanjayan 2006). Connectivity in the context of wildlife movements has two components: structural and functional connectivity (Tischendorf and Fahrig 2000b). The first component, structural connectivity, relates to the amount and spatial arrangement of habitat on the landscape. Functional connectivity incorporates a species' behavioral response to the structural landscape and is likely to differ among all but the most closely-related species (DEon et al. 2002). Connectivity among protected areas is a necessary component of any conservation planning or climate adaptation strategy (Griffith et al. 2009, Margules and Pressey 2000) and therefore lands outside protected areas are critical for maintenance of biodiversity (Franklin and Lindenmayer 2009). Unfortunately, land use change negatively affects landscape connectivity both by increasing the isolation of habitat patches and by decreasing suitable habitat in the matrix (Goodwin and Fahrig 2002). One of the primary strategies promoted to combat the effects of land cover and land use change leading to habitat loss and fragmentation is the creation, restoration, and management of corridors to maintain connectivity among protected areas (Beier and Brost 2010, Noss 1987). The effectiveness of corridors as a biodiversity conservation strategy has been questioned (Noss 1987, Simberloff et al. 1992), however, they have proven to be effective, at least in some applications (Damschen et al. 2006, Beier and Noss 1998, Gilbert-Norton et al. 2010, Haddad et al. 2003, Haddad and Tewksbury 2005). Given this, corridors are a key strategy for maintaining the resilience of biological systems to land use change and their adaptation to climate change. A well-connected landscape will facilitate the species movements and range shifts that are anticipated to be necessary as some species fail to thrive in their current range and need to colonize new areas in response to climate change to survive in nature (Griffith et al. 2009, Beier and Brost 2010).

When looking at connectivity among protected areas within a region, we are assessing them as a network. A network is simply a collection of units that may be interacting as a system (Proulx et al. 2005). Graph theory is a branch of mathematics that quantifies connectivity among nodes within a network (Urban et al. 2009). It has become widely used in ecology to model ecological networks and connectivity among protected areas and habitat patches (Saura and Pascual-Hortal 2007, Galpern et al. 2011). Functional connectivity takes into account the behavioral response of a species to a given landscape configuration and improves the quality of corridor design when compared with corridors based on simple Euclidean movement distances for species (Beier et al. 2008, Cerdeira et al. 2010). One approach is to combine graph theory with least-cost path analyses. This approach makes use of cost surfaces, which are raster surfaces that assign values indicating the difficulty of movement for a given species through each pixel based on, for example, land cover type. Cost surfaces can be used to generate least-cost paths, a raster of pixels likely to be used by a species for movement between habitat patches in a landscape because that path has a lower cumulative cost of movement than other paths across the landscape. For example, a short path composed of high-cost pixels may be less costly to traverse than a longer path of low-cost pixels. Least-cost paths differ from simple Euclidean distances in that the distance to cross a pixel is multiplied by the resistance assigned to crossing that particular land cover. Least-cost paths have the potential to provide important information on the connectivity within our existing protected area network from a species-specific perspective (Saura and Pascual-Hortal 2007, Theobald et al. 2006, Saura and Torne 2009). Combining least-cost paths with graph theory allows for incorporation of landscape structure and species behavior information into distance and probability of connectivity calculations (Laszczak et al., in review).

A survey of decisionmakers, scientists, and policymakers identified as a top priority the need to determine how changes in land use and climate will impact the effectiveness of protected areas (Fleishman et al. 2011). I have been working with managers and policymakers while developing three main questions to address this research need. The first two questions focus at the continental scale, while the third is regional in scope.

First, I will assess how future land use change affects the U.S. Fish and Wildlife Service National Wildlife Refuge System (NWRS) in the 48 contiguous United States.

Second, I will and assess how housing growth, specifically, affects the NWRS in the 48 contiguous United States.

Third, I will examine at a regional scale the effect of future land use and housing change on the protected area network in a specific ecoregion for species with limited dispersal and movement capability.

My dissertation will provide the first system-wide evaluation of the spatial distribution of land use changes and housing growth for a large protected area network, the National Wildlife Refuge System. I will provide important information that could be used in assessments of the resilience of the NWRS and will determine opportunities for land use and climate change adaptation by identifying existing and potential areas for corridor management/restoration and projections of future corridors.

Chapter 1

Current and Future Land Use and Land Cover around the U.S. Fish and Wildlife Service National Wildlife Refuges

Introduction

Protected areas are one of the main tools used to conserve biodiversity (Wiens 2009). However, the condition of the surrounding landscape is important to their effectiveness (Franklin and Lindenmayer 2009, Prugh et al. 2008). This is a concern since human land use surrounding protected areas is often intensive (Gaston et al. 2008, Joppa et al. 2008, Wade and Theobald 2010, Radeloff et al. 2010). Land use surrounding protected areas is a concern worldwide (Joppa et al. 2008, Hansen and DeFries 2007a). The intensity of land use in the surroundings has direct impacts on the ability of protected areas to meet their conservation goals because the surrounding land uses affect ecological processes (Hansen and DeFries 2007b). The potential for disruptions to ecological flows means that protected areas cannot be viewed in isolation but must be managed in the context of their surroundings (Wiens 2009).

The condition of matrix habitats, defined here as those areas outside protected areas, is important for the maintenance of biodiversity (Franklin and Lindenmayer 2009). However, humans have substantially altered the earth's ecosystems, appropriating many resources for human use (Vitousek and Mooney 1997)(Vitousek and Mooney 1997). The majority of the earth's land surface is now used for either human settlements or agriculture with less than 25% remaining in a wild state (Ellis et al. 2010). Land use patterns in the United States are comparable to the greater global patterns of ongoing conversion for human use (Leu et al. 2008, Theobald et al. 2009, Theobald 2010). Current changes in economic development activity to an information economy may exacerbate these changes (Huston 2005)(Huston 2005).

The conversion of land to human-dominated use has associated impacts on ecosystem process and biodiversity that can be the result of habitat loss as well as degradation of ecological processes (Flather et al. 1998, Foley et al. 2005b). Land use can impact biodiversity through both habitat loss and fragmentation of existing habitat (Fischer and Lindenmayer 2007). Current and historic land use with associated loss and fragmentation of habitat affects water quality (Locke et al. 2006, Allan 2004)(Locke et al. 2006, Allan 2004), community composition (Attum et al. 2008, Pidgeon et al. 2007, Lepczyk et al. 2008), dispersal and movements (Fahrig 2007, Mazerolle et al. 2005), and invasion by non-native species (Gavier-Pizarro et al. 2010, Predick and Turner 2008, Kuhman et al. 2010).

Land use changes beginning with agricultural production, followed by growth of population centers, and finally freedom from the constraints of primary production and industrialization are the typical path to human conversion of the landscape and are reflective of the pattern of land use change in the United States (Brown et al. 2005, Huston 2005). Forecasting land use change can be a powerful planning tool and is essential to effective conservation planning (Merenlender et al. 2009b, White et al. 1997). Methods have been developed that exploit the linkages within land use history to forecast land use change at an ecoregional level (Sohl et al. 2010). Such models can assess the likely impact of different economic policies and scenarios on future land use patterns (Radeloff et al. in review).

In the U.S. the only network of protected areas designed primarily for the protection of biodiversity is the National Wildlife Refuge System (NWRS) of the U.S. Fish and Wildlife Service (FWS). Other Federal lands (i.e. U.S. Forest Service, Bureau of Land Management, National Park Service) are typically managed for multiple uses. Additionally, the NWRS has a stated goal of maintaining the biological integrity of the system (Meretsky et al. 2006). This goal is endangered by the fact though that many of the refuges occur in a matrix of other intensive land uses (Scott et al. 2004). Given concerns that climate change will likely exacerbate current impacts to the NWRS, the FWS is interested in assessing future land use change as a component of climate change adaptation (Griffith et al. 2009). While there is a great deal of uncertainty about both the future impacts of climate change and land use, effective management necessitates the development of models at multiple spatial and temporal scales to inform planning processes (Heller and Zavaleta 2009, Burgman et al. 2005)(Heller and Zavaleta 2009, Burgman et al. 2005). This calls for evaluating the current state of land use surrounding the National Wildlife Refuges and developing scenarios of potential future land uses (Griffith et al. 2009, Meretsky et al. 2006, Scott et al. 2004).

I propose to evaluate current and future land use around the NWRS in the contiguous 48 United States. I will address two questions. First, what is the current land use in the areas surrounding the National Wildlife Refuges? Second, what is the likely distribution of land use under different economic and policy scenarios?

Methods

I will use the 2001 National Land Cover Dataset (NLCD) to map current land use and previously-developed econometric models to map future land use scenarios for the lands surrounding the National Wildlife Refuges across the United States.

Study Area

The NWRS consists of over 600 units and 37 wetland management districts. These refuges are unique in that they are typically small relative to other Federal lands and are embedded within a matrix of intensive land use including agriculture and residential development (Griffith et al. 2009, Scott et al. 2004). I will base my analysis on the FWS Cadastral Geodatabase, which includes all FWS property interests (http://www.fws.gov/GIS/data/CadastralDB/index.htm). I will narrow the analysis to include only properties in the conterminous United States and exclude properties in Alaska, Hawaii, and United States Territories. In addition, I will only evaluate properties that are designated as National Wildlife Refuges. This means that I will exclude properties that might receive similar management (i.e. National Game Ranges) but that are normally managed by other governmental units (e.g. states wildlife management agencies) through formal agreement. This will reduce the analysis to 494 properties (Figure 1). I will further limit my analysis up to 75 km around each of the refuge units that occur within the conterminous United States assuming that matrix habitat beyond that distance has limited effects on the refuges themselves (Fig. 1).

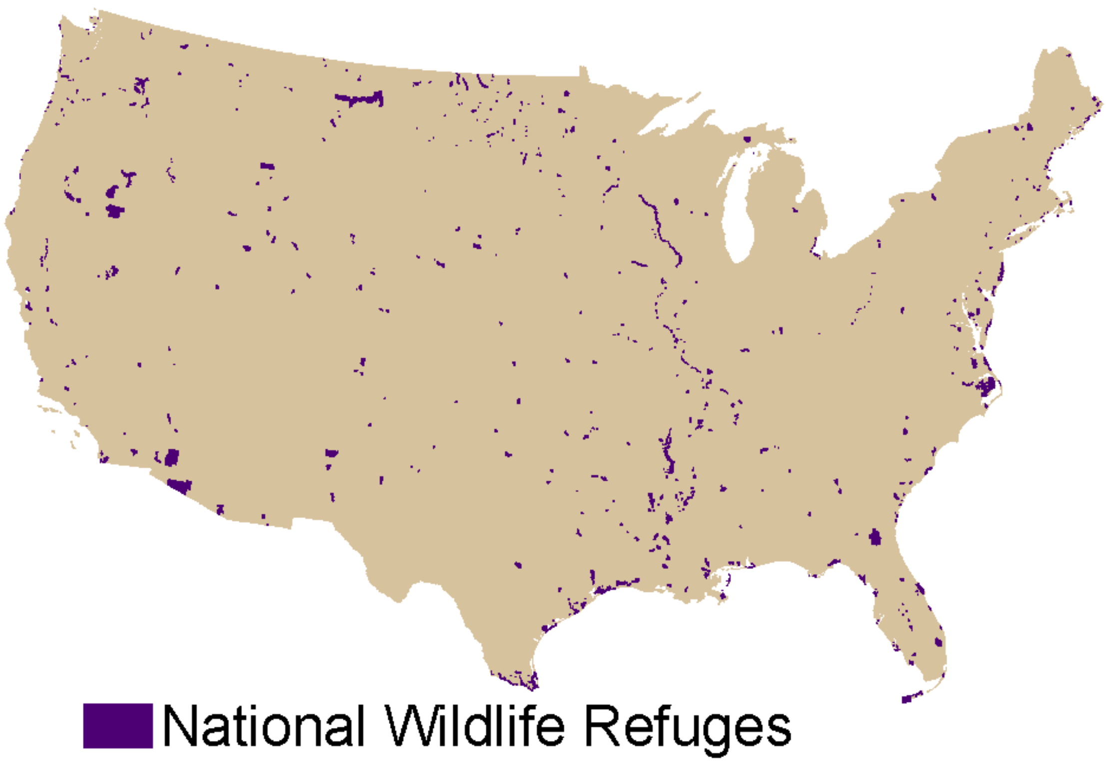

Figure 1. Map of the 494 National Wildlife Refuges used in the analysis of land use and housing change.

Land Use Data and Future Scenarios

Current land use will be determined using the 2001 NLCD. Future land use will be predicted using a set of previously-developed models based on economic and policy scenarios. The base model is based on data from the U.S. Department of Agriculture National Resources Inventory (NRI), which uses plot level data to model probability of change among different land uses (Radeloff et al., in review). The first policy, "business-as-usual", is a baseline scenario that simply continues the trends found between 1992 and 1997 in the NRI. The second scenario is an afforestation scenario where forestry receives increased value. The third scenario involves the removal of agricultural subsidies and the final scenario is based on increased urban land value due to population increase. The models will be used to predict land use transitions among five land cover types: urban lands, forest, grasslands, cropland, and rangelands out to the years 2030 and 2060.

Land Use Summary

I will limit my analysis to housing within 5-, 25-, and 75-km of National Wildlife Refuges in ArcGIS 9.3 (ESRI, Redlands, CA). The distances were chosen to conform to the original USFWS policy guiding habitat restoration on private lands which prioritized restoration projects that benefited the National Wildlife Refuges. Much of the early habitat restoration work focused on wetlands for waterfowl. The 5-, 25-, and 75-km distances equate to mallard brood movements (Mauser et al. 1994), diurnal winter movements of females (Davis and Afton 2010), and post-breeding pre-migration movements (Gaidet et al. 2010). Those distances also encompass home ranges and seasonal movements for a variety of species of interest to FWS, such as Blanding's turtle, sage grouse, and Canada lynx (Connelly et al. 1988, Grgurovic and Sievert 2005, Ruggiero et al. 1999). In addition, the distances will accommodate the fact that the model outputs use county-level data and are more accurate at broader scales. I will generate 5 raster maps at each buffer distance. The raster maps will represent the transition probabilities from an initial land cover to 1 of the 5 land cover types identified above. I will summarize the probability of transitioning to a particular land use among the categories under each economic/policy scenario, for each of the 494 refuges in the contiguous United States, in 2010, 2030, and 2060. I will create multiple replicates of each scenario during my analyses using ArcGIS. I will use the replicates to analyze the robustness of my results. Prior analyses with this data examined 500 replicates to determine robustness by randomly selecting simulations and plotting the net change in land use when compared with starting conditions, and found that about 100 replicates suffices to obtain robust results. The future land cover distribution is derived from the average of the replicates. The scale of the original econometric analysis included the entire United States, so the finer scale of my analysis within specified distances of National Wildlife Refuges may require a higher number of replicates to find the point at which net change will stabilize.

Based on recent analyses of nationwide trends, I anticipate that exurban land use growth and afforestation will be the primary land use changes around the NWRS, though loss of pasture and rangeland may also occur (Radeloff et al. in review).

Outcomes

I will produce a set of tables with the distribution of land use around each refuge of the NWRS at 5-, 25-, and 75-km distances for 2010, 2030, and 2060. In addition, I will be able to generate figures depicting transition among land uses for 2010, 2030, and 2060. I anticipate that impacts to refuges will differ based on their location in the U.S. landscape and that those impacts will manifest themselves differently at different scales of analysis.

Chapter 2

Current and Future Housing Distribution around the U.S. Fish and Wildlife Service National Wildlife Refuge System

Introduction

Conservation lands cannot be viewed in isolation but must be viewed in the larger context in which they occur (Wiens 2009). The condition of matrix habitats, defined here as those areas outside protected areas, is an important factor in the maintenance of biodiversity (Franklin and Lindenmayer 2009). Housing growth is a key threat to biodiversity in the United States (Flather et al. 1998, Hansen et al. 2005). Housing does not occur as an isolated phenomenon but is associated with infrastructure development activities which have their own environmental effects (Hawbaker and Radeloff 2004). Habitat changes related to housing and associated development such as roads affect individual species (Merenlender et al. 2009a), community composition across taxonomic groups (Pidgeon et al. 2007, Eigenbrod et al. 2008, Miller et al. 2003), predation rates (Wilcove 1985), species abundance and distribution (Fahrig and Rytwinski 2009), presence of invasive species (Gavier-Pizarro et al. 2010), and ecological flows (Hawbaker et al. 2006, Patrick and Gibbs 2010). In spite of the importance of housing, our methods for identifying the location of housing are imperfect and residential development often remains undetected (Pidgeon et al. 2007).

Native species diversity and abundance tends to decrease with increased housing density along the rural-to-urban fringe (Hansen et al. 2005). While this is generally true, low-density rural housing (densities ranging between 6 and 25 houses/km^2) has been a fast-growing land use in the United States and has its own associated problems (Brown et al. 2005). The footprint of this form of development is proportionately larger per housing unit when compared with suburban sprawl (Radeloff et al. 2005b). In addition, rural residential housing growth in recent decades has been faster near the boundaries of public lands than across the rest of the United States (Hammer et al. 2009). This form of development is driven by the amenity-rich nature of protected areas and this growth pattern is problematic because many of these areas are crucial for biodiversity conservation. In addition, housing is a particularly insidious form of land use because of its persistence. Once land is converted to housing, it is likely to remain in that use. The recent pattern of housing growth may thus be reducing the conservation value of these lands (Radeloff et al. 2010).

The higher rate of housing growth at the boundaries of protected areas has been well documented (Radeloff et al. 2010, Hammer et al. 2009, Leinwand et al. 2010). However, prior research has only reported average housing densities, and there has been little work done to evaluate the spatial distribution of housing at these boundaries (Gaston et al. 2008, Joppa et al. 2008). The spatial distribution of housing is important because comparable densities of housing within buffers may be configured differently, with some isolating refuges and others still maintaining some degree of connectivity and allowing ecological flows (Figure 2). The U.S. Fish and Wildlife Service (FWS) manages a National Wildlife Refuge System (NWRS) that is unique among U.S. Federal lands in that its focus is on wildlife conservation while other Federal lands (i.e. U.S. Forest Service, Bureau of Land Management, National Park Service) are managed for multiple uses. Additionally, the NWRS has a stated goal of maintaining the biological integrity of the system (Meretsky et al. 2006), which is complicated by the fact that many of the refuges occur in a matrix of other intensive land uses (Scott et al. 2004). Given concerns that climate change will likely exacerbate other stressors that include urbanization, habitat loss and fragmentation, FWS is interested in improving connectivity for the NWRS as an adaptation measure for climate change (Griffith et al. 2009). This requires evaluating the current state of the areas surrounding the National Wildlife Refuges in support of conservation planning and to acknowledge that the conservation value of land decreases with increased housing density (Hansen et al. 2005).

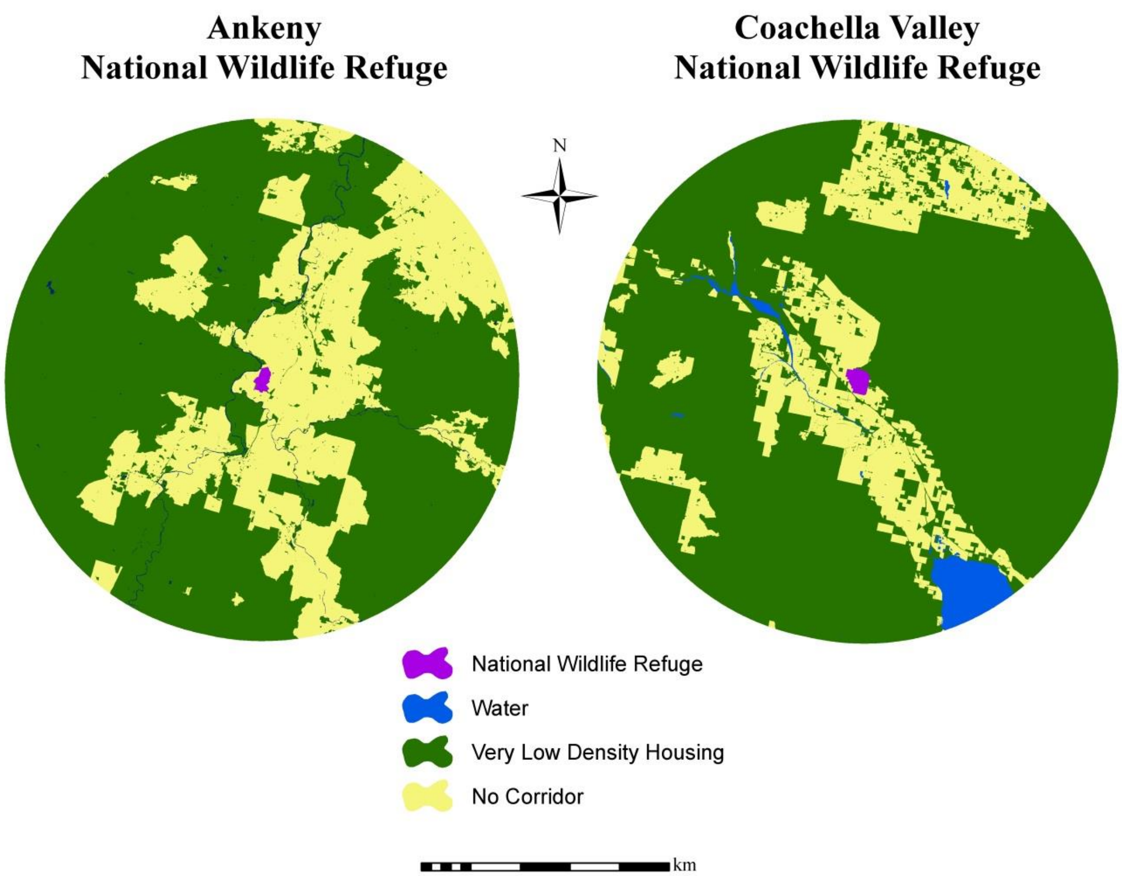

Figure 2. Two refuges with similar housing pressures at 50km but with different spatial distribution of the housing. Ankeny NWR does not directly connect to a corridor through very low density housing while Coachella Valley NWR does connect directly.

I propose to evaluate the spatial distribution of housing around the NWRS in the conterminous United States. I will address three questions with my research. First, does the current pattern of low housing density provide corridors or opportunities to restore corridors around National Wildlife Refuges across the United States? Second, what is the current distribution of wildland vegetation within any existing low housing density corridors? Third, how are these low housing density corridors likely to change in the future?

I predict that there will be many low housing density corridors but that the current land cover in these corridors is in some type of intensive use (e.g. row crops) requiring extensive restoration and management to improve their ability to function as wildlife habitat corridors for many species. I also predict that under future scenarios of housing growth the number and relative area of corridors will be reduced when compared with previous conditions, increasingly isolating the protected areas.

Methods

I will use previously-developed housing density data to quantify and map current and future low-density housing corridors for a set of protected areas across the United States (Hammer et al. 2009). Then I will use the National Land Cover Dataset in conjunction with the housing data to identify areas of existing habitat within areas of low housing density to quantify existing habitat and potential opportunities for management of corridor habitats.

Study Area

The NWRS consists of over 600 units and 37 wetland management districts (Figure 1). These refuges are unique in that they are typically small relative to other Federal lands and are embedded within a matrix of intensive land use including agriculture and residential development (Griffith et al. 2009, Scott et al. 2004). I will base my analysis on the FWS Cadastral Geodatabase, which includes all FWS property interests (http://www.fws.gov/GIS/data/CadastralDB/index.htm). I will limit my analysis to the same set of National Wildlife Refuges as in Chapter 1.

Housing Data

Housing data were obtained from the 2000 U.S. Decennial Census. The Census provides a count of all housing units in the United States. Housing growth rates during the 1990s were used to project future housing growth out to 2030 (Radeloff et al. 2010). Housing units were totaled by count for each decade and adjusted using county-level housing projections. County-level forecasts were created using 2008 Woods and Poole county projections (http://www.woodsandpoole.com/). Woods and Poole Economics, Inc. is a firm that does long-term forecasting using an annually updated database of county-level demographic, economic, and household data. The projections were derived using an advanced demographic model and are considered the best available population forecasts. County-specific household sizes were used to convert population growth to housing unit growth. Converting population size into housing density adjusts for high frequencies of vacant housing units in areas with high proportions of seasonal housing.

Housing Summary and Corridor Analysis

I will limit my analysis to housing within 5-, 25-, and 75-km of National Wildlife Refuges in ArcGIS 9.3 (ESRI, Redlands, CA). These buffer distances and the justifications for their use are identical to Chapter 1. I will apply the stated buffers to each of the 494 refuges in the conterminous United States, summarizing the number of housing units projected to occur at each buffer distance for the years 2010, 2030, and 2060.

I will conduct the low-density housing corridor analysis using ArcGIS 9.3 by creating contiguous polygons of low-density housing in the buffers that surround the National Wildlife Refuges. Polygons of low-density housing that touch both the National Wildlife Refuge boundary and the outer boundary of each buffer (i.e. "the outside world") will be designated as corridors. I will select areas with fewer than 6.17 housing units/km^2 as low-density housing. These areas are designated as either uninhabited or very low density according to wildland-urban interface definition (Radeloff et al. 2005a, USDA and USDI 2001). In addition, I will conduct a sensitivity analysis using 3.09, 12.34, and 18.51 housing units/km^2 from the 2010 housing data to determine how the choice of housing density level affects the presence of corridors. I will summarize the number of corridors, the frequency of occurrence of corridors, and the average proportion of buffer area that comprises corridors around refuges at all buffer distances and at each time step as well as for the sensitivity analysis.

Finally, I will use ArcGIS 9.3 to evaluate the current condition of habitat in low-density housing corridors. I will create masks of the 2001 National Land Cover Dataset (NLCD) within each identified corridor. I will summarize the amount of land in each of the NLCD cover classes at all three buffer distances. In addition, I will convert the raster dataset to a polygon dataset and determine whether any of the low-density housing corridors have contiguous polygons of wildland that touch on both National Wildlife Refuge boundaries and "the outside world" at each buffer distance. If so, these would constitute both low-density housing corridor and an existing habitat corridor for which I create a summary of number of true corridors and average proportion of buffer area that comprises the corridors. I will do this analysis in four ways. The first two will incorporate the NLCD open water class (class 11) as habitat paired with one analysis where hay/pasture (class 81) is considered habitat and a second where it is not. The second two analyses, with open water as non-habitat, will also be paired with an analysis where hay/pasture is considered habitat and a second where it is not. All other land cover classes will be the same among analyses. I will define the rest of wildlands as the following NLCD landcover classes: deciduous forest (class 41), evergreen forest (42), mixed forest (43), scrub-shrub (52), grassland/herbaceous (71), woody wetlands (90), and herbaceous wetlands (95). This excludes developed lands (classes 21 through 24), barren lands (31), and cropland (82).

Outcomes

I expect to produce a set of tables with the number, size (relative to buffer area), and frequency of occurrence of low-density housing corridors around the NWRS as well as a tabulation of the land cover classes within the low-density housing corridors. In addition, I will create two sets of tables for the number, proportion of land area in corridors (relative to buffer area), and frequency of occurrence of low-density housing corridors for 2030 and 2060 based on housing growth projections. Finally, I expect to be able to generate figures depicting existing conditions of housing distribution and low-density housing corridors for 2010 as well as projected housing conditions for 2030 and 2060.

Chapter 3

Current and Future Connectivity among Protected Areas for Three Species with Limited Dispersal Capability in the Northern Hardwoods Ecoregion of the Upper Midwestern United States

Introduction

One of the primary strategies promoted to combat the effects of land cover and land use change leading to habitat loss and fragmentation is the creation, restoration, and management of corridors to maintain connectivity among protected areas (Beier and Brost 2010, Noss 1987). While there has been debate about the effectiveness of corridors as a conservation strategy, reviews of the literature and empirical studies have demonstrated the conservation value of corridors (Damschen et al. 2006, Noss 1987, Simberloff et al. 1992, Beier and Noss 1998, Gilbert-Norton et al. 2010, Haddad et al. 2003, Haddad and Tewksbury 2005). Corridors are probably the primary tool promoted to provide resilience within biological systems and are a key component of climate change adaptation strategies. It is anticipated that species movements and range shifts will be necessary for the continued survival of some species as they fail to thrive in their current range and need to colonize new areas in response to climate change. The hope is that a well-connected landscape will reduce impediments to these movements (Griffith et al. 2009, Beier and Brost 2010). With limited funding, it is critical that such investments incorporate an assessment of threat in order to maximize conservation gains (Merenlender et al. 2009b).

Connectivity has been defined as the degree to which a landscape facilitates or impedes movements among resource patches (Taylor et al. 1993). Landscape connectivity is affected by both habitat loss (i.e. overall reduction in the amount of habitat) and habitat fragmentation (i.e. the breaking apart of habitat) with habitat loss having consistent negative impacts to biodiversity, while habitat fragmentation effects are weaker and more variable (Fahrig 2003). In addition, connectivity has two components: structural and functional connectivity (Tischendorf and Fahrig 2000a). Structural connectivity is the amount and spatial arrangement of habitat on the landscape. Functional connectivity recognizes that the same landscape will have relatively different connectedness for different species, incorporating species behavioral response to the landscape. The second component, functional connectivity, can also be divided into potential and actual connectivity. Potential functional connectivity is simply an indirect measure of functional connectivity (i.e., based on a model), while actual functional connectivity is quantified based on actual species movements (Crooks and Sanjayan 2006).

I propose to evaluate the change in potential functional connectivity (i.e., modeled rather than quantified via telemetry or genetics) for a network of protected lands in the upper midwestern United States. I will use the structural connectivity outputs from the Chapter 1 and Chapter 2 analyses to identify plausible future land use scenarios on which to base the potential functional connectivity analysis. Additionally, I will examine the implications of the modeled landscapes for three species from different taxonomic by addressing the following questions. First, what is the degree of functional connectivity in the current landscape for each of the three species? Second, how does functional connectivity change under the 2030 and 2060 future land use scenarios from Chapter 1? Third, how does functional connectivity change by 2030 and 2060 given the future housing growth projections from Chapter 2? Finally, what are the functional protected area networks in 2010, 2030, and 2060 under the three policy scenarios from Chapter 1 and projections from Chapter 2?

I hypothesize that the degree of landscape connectivity will decrease over time. In addition, I predict that the number of functional networks will increase while the size of the networks decreases over time because larger functional networks will be broken down into smaller, more numerous networks. However, I also hypothesize that the networks and connectivity will differ among species. I predict that species that are more sensitive to infrastructure development such as roads will experience a higher rate of connectivity loss and habitat network fragmentation over time than species which are more tolerant of that sort of landscape change.

Methods

I will use Blanding's turtle, Northern Leopard Frog, and the American Badger as model species for this simulation. I anticipate that they will provide a diverse response to changes in landscape connectivity and structure. While they all have limited dispersal capability, the differences among physiologies, habitat requirements and sensitivity to landscape features such as roads or cropland will cause the landscapes to vary in suitability among the species.

Study Area

I will create models of habitat and protected area connectivity for these species within the North Central Hardwood Forest Ecoregion (Omernik 1987) (Fig. 3). This ecoregion is largely composed of a mosaic of forest, wetland, and agricultural land covers that include a mix of cropland, pasture, and dairy. It is the transition zone between the forested landscape of the Northern Lakes and Forests ecoregion to the north and the agricultural regions of the Central USA Plains and Temperate Prairies to the south and west (Omernik Level II). I will evaluate all protected lands from the U.S. Protected Area Database with GAP protection status 1, 2, and 3 within this region (http://gapanalysis.usgs.gov/data/padus-data/).

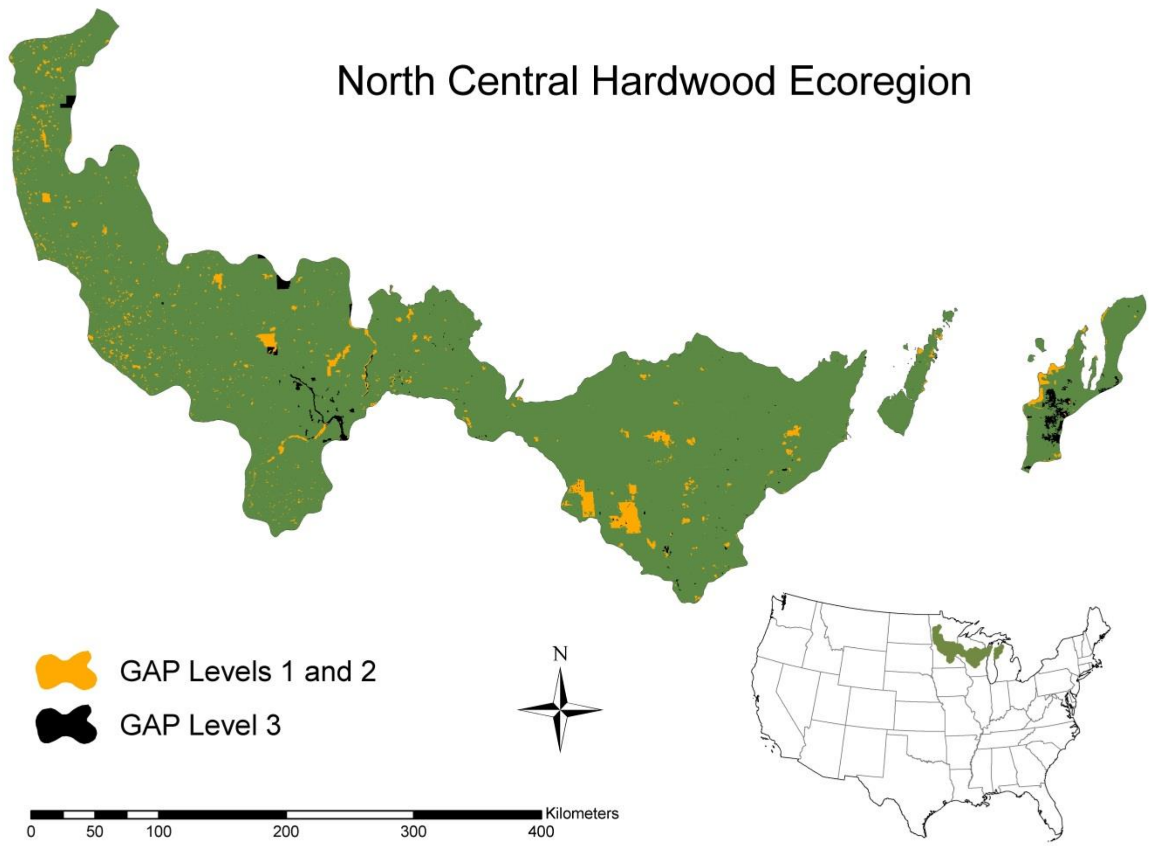

Figure 3. Protected areas within the North Central Hardwood Ecoregion (Omernik) from the GAP Protected Area Database. GAP Level 1 includes areas with permanent protection where ecological disturbance events are allowed to occur, Level 2 includes areas with permanent protection where ecological disturbance is often suppressed, and Level 3 includes areas with permanent protection but subject to extractive use or off-highway vehicle activity.

Habitat Mapping

I will determine suitable habitat for each of the species using MaxEnt. MaxEnt is a machine learning program that is capable of using presence-only species records to model species distributions and generally outperforms other programs (Phillips et al. 2006). I will use known locations from Natural Heritage Inventory databases to determine presence. I will use MaxEnt to determine the most important predictors of occurrence among the variables including habitat patch size, land cover class from the 2001 NLCD, density of roads near known occurrences, housing density near known occurrences, soil type, seasonal rainfall, and seasonal temperatures. I will use the best performing model for each species to generate habitat suitability maps for each species. Finally, suitable habitat patches will be divided into three categories based on size: patches suitable for breeding populations, patches suitable for breeding, and patches suitable for use by individuals (Beier et al. 2008).

Least Cost Paths

I will use land cover resistance values determined from published peer-reviewed literature and, where the literature is insufficient, I will seek out expert opinion to develop a cost surface for movement among habitat patches for each of the three species. I will then develop cost-distance surfaces for each species using FunConn, an ArcGIs-compatible functional connectivity modeling tool for ArcGIS (Theobald et al. 2006). FunConn is able to use the outputs I will generate with MaxEnt and will generate cost-distance values among habitat patches that will be useful in subsequent analyses of landscape connectivity. I will buffer each path at distances of 250 and 500 m to determine corridors potentially available for use by each species.

Landscape and Network Connectivity

I will use Conefor Sensinode 2.2 to quantify landscape connectivity for each species. It uses graph theory to generate landscape connectivity indices. Specifically, it generates a Probability of Connectivity Index which outperforms the many other connectivity measures that are not bounded between 0 and 1, do not indicate decreases in connectivity when portions of patches are lost, and do not consider the loss of larger patches as more important than the loss of smaller patches. In addition, it generates relative importance-to-connectivity values for specific habitat patches, which can be very useful for prioritizing conservation actions (Saura and Torne 2009). I will use the outputs of each habitat suitability model and the cost distances from MaxEnt and the Least-Cost-path analysis as inputs for Conefor Sensinode to determine baseline connectivity of the Northern Hardwood Forest landscape for each of the species in 2010.

In addition, I will determine network connectivity using Igraph, a software package that uses graph theory to create network graphs (http://igraph.sourceforge.net/introduction.html). Graph theory is a mathematical concept that assesses connectivity, routing and flow within networks and can be used to model landscape mosaics (Urban et al. 2009). The package can be used to generate graph metrics such as minimum spanning trees, network flows, network components and other measures that I will use to quantify changes in networks under different future scenarios. Igraph can be used in both the R statistical and ArcGIS software environments.

Change Over Time

Finally, I will examine potential changes in functional connectivity for each species in the future. I will use the outputs of the housing change and land use change from Chapters 1 and 2 within the North Central Hardwood Forest ecoregion to determine projections and scenarios of change within the corridors present on the landscape in 2010 for the years 2030 and 2060. I will then re-calculate landscape and network connectivity using the Conefor Sensinode 2.2 and Igraph software packages for each species in 2030 and 2060.

Outcomes

I expect to produce graphics depicting changes in connectivity over time for each of the three species that are likely to result from both future housing growth (from housing projections) and future land use change (from land use change scenarios). The graphics will depict changes in functional corridors at both 250 and 500 m buffer distances. In addition, I will generate graphics depicting likely changes in protected area network connectivity for all three species at all three points in time resulting from both housing and land use change. Finally, I will quantify values of landscape connectivity metrics, importance of specific protected areas to connectivity, numbers of functional networks, and average size of functional networks for each species at each point in time.

Overall Significance of my Dissertation

The USFWS NWRS is one of the world's largest protected area systems (Griffith et al. 2009). It is an excellent model for my analyses where I am working with policy makers and administrators at the national scale and with refuge planners and resource managers at the regional scale. Research and planning efforts like this have been criticized for generating information that is either not useful to or never used by resource managers. This has been called "the research-implementation gap" (Knight et al. 2008). The coordination of my dissertation with implementers will bridge this gap and provide specific, useful information for stated planning and management needs.

The intensive use of land surrounding protected areas for either agriculture or residential development is a serious threat to conservation and our protected area network (Wade and Theobald 2010, Radeloff et al. 2010). My work will have direct application to protected area management for the NWRS and for protected lands within the North Central Hardwood Forest ecosystem by addressing a major concern in protected area management, evaluation of the condition of surrounding lands. This is critical to the ability of NWRS lands to fulfill their conservation mission because surrounding lands influence the effective conservation area and connectedness of protected lands (Wiens 2009). I will complete analyses that provide current and future assessments of habitat connectivity within the region with respect to housing and land use.

In addition to these concerns, our protected area network faces new challenges in the combination of existing stressors (i.e. habitat loss and fragmentation, invasive species, pollution, etc.) with global climate change (Heller and Zavaleta 2009, Griffith et al. 2009)(Heller and Zavaleta 2009, Griffith et al. 2009). A major current theme for conservation is climate change adaptation, which often includes the concept of landscape connectivity to promote ecological resilience. Adaptation requires identifying the what, who, and how for any adaptation strategy as well as analyses of possible futures that incorporate larger spatial and temporal resolution (Heller and Zavaleta 2009, Smith et al. 2000)(Heller and Zavaleta 2009, Smith et al. 2000). While adaptation is mostly spoken of in terms of climate change, the strategy is equally applicable to the process of land use change. My analyses of land use and, more specifically exurban residential land use will provide a solid foundation for adaptation to future change.

In addition to the management implications, my work will provide a unique look at the spatial distribution of housing and changes in land use in relation to a system of protected lands. Previous work has been done that indicates housing growth occurs at a higher rate near protected areas but that work has not explicitly addressed the spatial distribution of that housing nor has it addressed resulting future changes in functional connectivity in a landscape (Wade and Theobald 2010, Radeloff et al. 2010, Hammer et al. 2009). My work identifying areas of existing corridors, potential areas for corridor management/restoration, and future projections of corridors using housing and land use change data in a functional framework will be a new contribution to our understanding of likely future changes in landscape connectivity. I will link a number of tools and models to identify potential future scenarios for conservation and protected area management through my projections of land use surrounding the NWRS and my development of corridor models in the Northern Hardwoods based on those projections.

Finally, my work will address the coupled human and natural system surrounding National Wildlife Refuges. Coupled human and natural systems are, defined as those systems that incorporate interacting natural and human mechanisms and processes (Liu et al. 2007b). My study incorporates four features of coupled human and natural system studies: multi-disciplinary research, large temporal scale, tools and data from social and ecological sciences, and addressing interactions between human and natural systems (Liu et al. 2007a).   Residential development and land use change are social phenomena and habitat corridors are an ecological concept. Studying residential development and land use change in combination with corridors to promote ecological flows acknowledges and integrates the linkages between these system components. The temporal scales at which we will be evaluating these changes will also contribute to our understanding of biodiversity conservation. These analyses will address the concept of ecological resilience in human-dominated systems.

Perhaps most importantly, my dissertation work will have broad management implications. I am conducting my analyses at multiple spatial and temporal scales. My analyses of a nationwide system of lands, connectivity within an ecoregion, and assessment of multiple stressors at multiple time steps will have implications for protected area management everywhere. It has the potential to provide important information to local managers and policy makers as well as methodological and theoretical contributions to the global conservation community.

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