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I, Teri A Jacobs, hereby submit this original work as part of the requirements for the degree of Doctor of Philosophy in Geography.

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Conservation Matters: Applied Geography for Habitat Assessments to Maintain and Restore Biodiversity

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Conservation Matters: Applied Geography for Habitat Assessments to Maintain and Restore Biodiversity

A dissertation submitted to the

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In partial fulfillment of the

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Doctor of Philosophy

In the Department of Geography

by

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ABSTRACT

The Earth stands on the precipice of the sixth mass extinction. This extinction risk facing half of all terrestrial life has triggered a growing crisis and the urgent need to save the world's biodiversity. In response, to conserve biodiversity, we need to know the spatial and temporal changes of the species, and how these changes are related to the physical environment.

This dissertation research was undertaken with this in mind—to benefit the conservation community, either through the delivery of accessible biogeographic methods or information to further the restoration or maintenance of biodiversity. Preserving the structure of the ecosystem is the best way to reduce biodiversity decline, and by preserving its structure, we preserve its services upon which we depend. We therefore need simple but efficient methods to *quickly* identify threatened areas. This is extremely important considering the accelerated rates of biodiversity loss and extinction.

As a primary goal, this dissertation endeavored to fill those research gaps and offer the conservation community some simpler and more effective *useful* and *usable* geospatial techniques for biodiversity conservation analyses. Secondary goals of the research were (1) to contribute to specific conservation programs for critically endangered species, (2) to inform about the status of habitat, and (3) to address top conservation research priorities. While not a specific objective, the research outcomes may influence public policy.

This three-article dissertation introduces two novel techniques: (1) development of a habitat suitability model in ArcGIS (ESRI 2017. ArcGIS Desktop: Release 10.2. Redlands, CA: Environmental Systems Research Institute) using kernel density estimation and a mortality-risk weighting factor on road density, the delimiting variable; and (2) a rapid hybrid change detection

technique using ENVI's SPEAR Vegetation Delineation tool (Exelis Visual Information Solutions, Boulder, Colorado) for classifying live green vegetation and ArcGIS to compare and quantify changes in time. For the latter, two studies incorporated the change detection technique. The pilot study performed the change detection with color-infrared aerial photography, while the follow-up investigation tested the feasibility of the method to handle high resolution multi-sensor data, given the difficulty obtaining data from the same or similar sensors. These studies represent the first of their kind.

This dissertation research provides widely *applicable, practical, and employable* geospatial models to perform habitat assessments for biodiversity conservation. Considering the expertise problems adopting Geographic Information Systems and remote sensing for ecological modeling, the easy-to-implement techniques introduced here for the conservation community to perform habitat suitability and change detection analyses fills a pressing research gap. Tailoring the dissertation research to management needs is another significant step in bridging the gap between geospatial specialists, ecology, and the conservation community.

The research also contributes practically to two current conservation programs: (1) the habitat suitability modeling identified priority areas where potential reintroduction of critically endangered and extirpated red wolves into the Daniel Boone National Forest may occur and (2) the change detection analyses showed where and how much change (loss) had occurred in the endangered southwestern willow flycatchers' critical riparian habitat in Mesquite, Nevada. Managers and decision-makers within the U.S. Fish & Wildlife Service and the Bureau of Land Management, respectively, can use this pertinent information to advance their initiatives. Dissemination of this information enables the timely development and implementation of solutions.

These findings are important, not only for adding to the body of knowledge about specific habitat suitability or changes, but also because of the implications for practice. Restoration of wildlife first requires an understanding of the habitat criteria that shape the distribution, abundance, and persistence of species, and we cannot stem habitat loss without first monitoring and documenting habitat changes and the factors influencing the changes.

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To my children, Arden and Emory, during all my time in the Master's and Doctoral programs, I hope I exemplified hard work, academic achievement, and most importantly following your dreams. I wish Arden luck in her pursuit of becoming a neuropsychiatrist and Emory in his pursuit of becoming an ichthyologist or his fallback job as a professional fisherman and guide.

To Shane Mitchell, I dedicate this dissertation. He "forced" me to take hiking, kayaking, and backcountry backpacking/hammock camping trips. It was during one of those trips that I

determined how I could use the data in a meaningful way, and it was because of those trips I found the energy and drive to get it all done!

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CHAPTER ONE

Geography, with its spatial and interdisciplinary perspectives, is the conservation discipline of the future. (Brown, 1999, p. 234)

INTRODUCTION

How to Conserve Biodiversity? A Biogeographical Question

The Earth stands on the precipice of the sixth mass extinction. The ever-increasing rate of extinction, currently estimated between 1,000 and 10,000 times greater than the background extinction rate, means that we lose dozens of species every day and hundreds, perhaps thousands, every year. For the first time in Earth's history of life, a mass extinction will have a biological cause: humans. Humans have already caused the extinction of 5-20% of the species (Chapin et al. 2000), and, with impending climate change, urbanization, and other anthropogenic activities, scientists expect biodiversity to decrease dramatically. This extinction risk facing half of all terrestrial life has triggered a growing crisis and an urgent need to save the world's biodiversity (Grehan 1993).

Biodiversity refers to the innumerable richness and variation of the living world, ranging from genetic variability within a species through the diversity among species and populations to the variety of communities, ecosystems, and landscapes (Orians and Groom 2006). In short, biodiversity encompasses the total sum of all living things. Biodiversity plays an important functional role in ecosystems, underpinning ecological functions and services, such as clean air and water, fertile soils, pollination, and pest/disease control, and contributing to ecosystem processes, such as primary production, nutrient flows, soil formation, and climate regulation (Orians and Groom 2006; Norris 2012; Corbane et al. 2015). Even the traits of individuals affect

ecosystem processes, functions, and services (Norris 2012). Beyond intrinsic value, biodiversity is fundamental for human well-being because we benefit from the direct provision of ecosystem goods and services (Orians and Groom 2006; Andrew et al. 2014), and the loss of biodiversity, essentially the life support system, threatens human populations (Diaz et al. 2006).

Habitat modification, fragmentation, degradation, and destruction represent the main drivers of biodiversity decline and loss (Collinge 2001; De Leeuw et al. 2002; Groom and Vynne 2006). Other direct and indirect human activities, such as overexploitation, introduction of invasive species, pollution, and climate change, also imperil biodiversity (Groom 2006). As the human population expands exponentially, the impacts on the environment intensify, and, while civilizations have benefited for centuries over the conversion of natural ecosystems to human-dominated systems and the exploitation of resources, the societal consequences of biodiversity loss can be costly and wide-ranging (Chapin et al. 2000). The management of invasive species, for example, ranges from US$1 billion to $137 billion annually, and the reductions in the supplies of water, food, fuel, structural materials, medicines, and useful species contribute to rises in consumer prices (Chapin et al. 2000). In general, the costs of remedial measures far outweigh the costs of preventative measures (Pagiola and Platais 2016). Efforts to address and prevent biodiversity loss, therefore, need to be substantially strengthened if ecosystem health and integrity will be maintained. Human well-being and welfare, and possibly survival, depend on it.

While conservation plans have focused primarily on single species to address biodiversity losses, the innumerable amount of biodiversity necessitates the use of time- and cost-effective, large-scale approaches (Franklin 1993). Thus, we need a more holistic approach to maintain or restore biodiversity and species conservation—hence, an ecosystem approach. Ecosystems, along with individual species, must be evaluated if a substantial portion of biodiversity is to be

conserved (Grehan 1993; Franklin 1993; Walker, 1995; Noss et al. 1995). All the interrelated living and the non-living elements interacting in a given area constitute the ecosystem (Odum 1971). For conservation purposes, the ecosystem refers to discrete entities, such as a vegetation type, plant association, natural community, or a habitat defined by ecological or geographical factors, to identify, classify, delineate, and map (Noss et al. 1995). The loss of any of these entities, whether in terms of quantity or quality, contributes to the loss in biodiversity. In turn, biodiversity influences ecosystem resilience and resistance to environmental change (Chapin et al. 2000).

Biodiversity Conservation Science

Considering the lack of knowledge on the numbers and distributions of species in existence (Richardson and Whittaker 2010), the ecosystem approach is the *only* effective way to preserve the mass of biodiversity—including the smaller, "lesser" organisms—and processes in poorly known or unknown habitats and ecological subsystems (Franklin 1993; Walker 1995). By protecting and restoring ecosystems, we maintain the various environmental functions and forms of habitats that biotic communities require for survival and thus promote ecosystem integrity and biodiversity. Ecosystem conservation, furthermore, directly addresses the primary causes of species declines: habitat loss, degradation, and fragmentation (Noss et al. 1995).

The ecosystem approach provides a valuable framework in biodiversity conservation science, integrating strategies to promote the protection of the spatial patterns, processes, and functions that meet the survival needs of all the species, rather than one focal species. However, how can we characterize the species' ecological requirements without studying every single species within an ecosystem or landscape? The concepts of the umbrella species, keystone species, and indicator species may achieve a compromise between the single-species approach

and the ecosystem approach and serve as a surrogate for broader habitat conservation (Lambeck 1997; Payton et al. 2002). Protecting an umbrella species indirectly protects all other species within the same habitat range as the umbrella species, thereby conservation operates on the landscape or ecosystem level. A keystone species is defined as a species upon which other species in an ecosystem largely depend and the removal of which would cause cascading change effects in structure, function, or diversity of a community (Payton et al. 2002). By protecting a keystone species, we protect the functional role of that species and maintain ecosystem integrity. An indicator species reflects the condition of the environmental condition and its rising or falling status can be used as a proxy to manage a habitat or ecosystem.

Regardless of the approach, the tenets within the disciplines of biology and ecology have traditionally informed *in-situ* conservation (on-site strategies to halt declines and manage species within the natural habitat) or *ex-situ* conservation (off-site strategies to preserve a species outside its natural habitat, such as in a zoo or within a seed bank, for potential restoration or reintroduction) (Brown 1999). These applied principles include, but are not limited to, species-area relationship, minimum viable populations, genetic erosion from small populations, competition, optimization theory, evolutionary stable strategy, r/K selection theory, niche theory, and patch dynamics (Caughley 1994; Whittaker et al. 2005).

Ecosystems are *spatial* systems at their core (Bailey 1996). Earth's climate and surface processes shape ecosystems and influence the biological processes and biota in any given location, and these geochemical processes vary by location. As a result, species diversity, richness, composition, and distributions exist along geographic gradients (Lomolino et al. 2010). Measures of diversity depend strongly on spatial scale (Brown 1999; Willig et al. 2003). In general, biodiversity decreases as distance from the equator increases and as elevation and

marine depth increases. The greatest diversity occurs at ecotones, transitional areas of vegetation between two different plant communities, such as riparian systems (Bailey 1996). Our understanding of species distributions, patterns of biodiversity, geographic ranges, and habitats, albeit inadequate, elucidates on where species can and cannot live (and why) and facilitates conservation planning. Given the spatial influence on biodiversity, geography sits in a unique position to address the problem of biodiversity conservation (Brown 1999).

The Role of Geography and Geographical Techniques in Biodiversity Conservation

To conserve biodiversity, we need to know the spatial and temporal changes of the species, and how these changes are related to the physical environment. These analyses require tools capable of handling the multitude of different plants, animals, habitats, and ecosystems occurring in different geographic locations, and the question on how and where to save biodiversity is, in essence, biogeographic (Grehan 1993).

Biogeographical theories have revolutionized biodiversity conservation science. Describing, explaining, and predicting patterns of biodiversity lie at the core of biogeography, from which conservation priorities and protected area schemes draw inspiration (Whitaker et al. 2005). At the forefront, MacArthur and Wilson's equilibrium theory of island biogeography has provided insight to nature reserve design (Lomolino et al. 2010) and sparked one of the most heated debates in conservation: the SLOSS (single large or several small) debate. Concepts of metapopulation dynamics, nestedness, and habitat corridors in conservation planning also sprang from geographic thought (Brown 1999; Kupfer and Malanson 2004; Lomolino et al. 2010). Success in conserving biodiversity hinges upon understanding the geographic information on species, from knowing where we should locate nature reserves or to where species will spread under changing climatic conditions (Lomolino et al. 2010).

This knowledge can be extracted through the use of geospatial data and tools. With advances in Geographic Information Systems (GIS) and remote sensing (RS), spatial knowledge has mushroomed into conservation-related fields (Zimmerman 2000). GIS and RS are commonly used techniques in geography. Geographers can link spatial patterns to vegetation dynamics or ecological processes, such as succession, energy flows, trophic webs, pollinator movements, and species migration, and identify ecological indicators to facilitate the management and monitoring of biodiversity (Kupfer and Malanson 2004). This ability to monitor biodiversity along with the environmental impacts of human activities and other threats can optimize design and management strategies (Pettorelli and O'Brien 2014). Even with the poor availability of biological data, geographers can apply satellite imagery or abiotic environmental classifications to remote mapping for regional conservation planning (Ferrier 2002).

For the geographer, maps are indispensable, essential, and the preeminent means to record and convey information about the spatial characteristics of a place. No graph, chart, spatial statistic, or words can compare to a map's ability to elucidate the complexities of spatial relationships or be understood at a glance. Most people can understand a map; much fewer can comprehend a scientific paper (Jenkins et al. 2011). Maps derived from aerial photography, the longest-available, temporally continuous, and spatially complete record of landscapes and landscape change, have routinely assisted ecosystem management and decision-making (Cohen et al. 1996; Morgan et al. 2010).

Maps produced in GIS serve as pivotal tools for many governmental and non-governmental organizations (NGOs). For example, the Gap Analysis Program (GAP), one of the most well-developed conservation programs, gathers spatially-explicit biophysical data and disseminates information on the conservation status of species, habitats, and protected areas

(Bailey 1999; Kupfer and Malanson 2004). The GAP datasets provide the means to map and analyze landcover, protected areas, and species distributions in the United States for long-term maintenance of biodiversity. GIS mapping and modeling for biodiversity conservation spans a wide array of applications. Researchers have utilized the power of GIS for population viability analysis (e.g., Akçakaya et al. 1995), endangered species management (e.g., Liu et al. 1995), endangered species restoration (e.g. Mladenoff and Sickley 1998), disturbance effects on species (e.g., Willson et al. 2003), predicting climate change impacts on species distribution (e.g., Iverson and Prasad 1998), predicting invasive species spread (e.g., Johnson and Padilla 1996), biodiversity modeling (e.g., Salem 2003), identifying indicator variables for monitoring biodiversity (e.g., Noss 1990), and habitat suitability (e.g., Hirzel and Le Lay 2008).

Using a series of simple GIS analyses, Jenkins et al. (2011), created a conservation success story. The first step of the hierarchal analysis revealed the highest concentration of threatened birds in the Americas within the Atlantic Forest. Secondly, nested within the Atlantic Forest, the state of Rio de Janeiro supported the highest concentration of the threatened birds, and, finally, within Rio de Janeiro, ReBio União stood out as the highest priority forest fragment for conservation efforts to prevent bird extinctions. Implementation of their GIS study findings led to the restoration of landscape connectivity with habitat corridors. With freely-available Google Earth satellite imagery, the public could observe the recovery process of the forest in the corridor, which the authors believed help emphasize the geographical transparency of the effort.

While the scientific literature evinces the wide use of GIS for conservation decision-making, the direct use of remote sensing (RS) for such remains limited, despite the fact RS acquires a large amount of geospatial information (Palumbo et al. 2016). RS data deliver sought-after details on habitat quantity and quality for conservation management (Mairota et al. 2015).

Conservation scientists, however, have failed to take full advantage of RS for the following reasons: it consists of a larger volume and more complex data than standard GIS data; prohibitive cost of data acquisition and software; restricted access to the most beneficial high resolution data; the lack of trained, skilled analysts within the conservation community; and the lack of investment by conservation organizations and institutions in building the capacity of RS (Pettorelli and O'Brien 2014; Buchanan et al. 2015; Palumbo et al. 2016). Yet, the potential for RS to support natural resource, environmental, and wildlife management is considerable (Turner et al. 2003; Pettorelli and O'Brien 2014; Andrew et al. 2014; Buchanan et al. 2015; Mairota et al. 2015; Willis 2015; Palumbo et al. 2016).

This potential use of RS to monitor environments, environmental parameters, habitat and species distributions, and conservation status as well as to evaluate management programs for effectiveness has become a prominent research topic (Pettorelli et al. 2014; Corbane et al. 2015). Successful conservation programs must be based on understanding the spatial distribution and change in distribution (Collinge 2001; De Leeuw et al. 2002). Ecological investigation to understand organisms and their environments has traditionally relied upon ground-based observations. While these data have high accuracy, the collection is labor-intensive, costly, and impossible in some remote or harsh environments. RS provides a more practical means to gather relevant information over larger scales (Kushwaha and Roy 2002; Alpin 2005). For species active during the day, RS can even assist in wildlife censuses, either through direct observation derived from high resolution imagery or tracking animals with radio collars and aircraft (Kushwaha and Roy 2002).

Over the last 30 years, the utility of RS to inform the quality of and stressors on biodiversity at all spatial scales has increased (Pettorelli and O'Brien 2014). Several reviews

have described in depth this utility of RS to ecology and conservation (see e.g., Kushwaha and Roy 2002; Turner et al. 2003; Kerr and Ostrovsky 2003; Alpin 2005; Wang et al. 2010).

Kushwaha and Roy (2002) highlighted RS techniques for wildlife habitat inventory and mapping; biotic and abiotic surface features mapping (e.g., vegetation composition, density, landforms); extent of habitat, distance to other critical habitat, and habitat corridor measurements; change detection; and habitat and breeding site predictions. In the most frequently cited review, Kerr and Ostrovsky (2003) organized the RS applications into three broad groups: (1) *landcover and land use classification*, from which derived wildlife habitat models, species distribution predictive models, presence/absence in habitat range, and landscape heterogeneity (i.e., biodiversity proxy); (2) *integrated ecosystem measurements*, such as the Normalized Difference Vegetation Index (NDVI) to calculate net primary productivity, differentiate between natural and human settings, detect land cover changes, and to serve as an indicator for landscape heterogeneity, biodiversity, and habitat suitability as well as surface brightness temperatures to determine energy efficiency of ecosystems and changes to energy budgets from disturbance; and (3) *change detection* to understand disturbances from human activity, such as habitat loss from deforestation, natural stochastic events such as wildfires, and climate change effects.

Turner et al. (2003) described two approaches using RS for conservation purposes: direct and indirect. With the direct approach, RS can detect organisms, species assemblages and communities. Analysts can use these measures to determine species composition and land cover, and, in combination with information on known habitat requirements, possibly produce precise estimates of potential species ranges and patterns of species richness. Advances in RS technology, such as hyperspectral and hyperspatial, enable RS sensors to obtain more

information and enhance their capabilities. We can monitor whales from space or extract leaf-surface and edaphic parameters. On the other hand, the indirect approach uses environmental variables—e.g., chlorophyll, soil moisture, phenology, topography, canopy structure—as proxies for analyzing primary production, climate, or habitat structure. We can apply climatic variables to predict areas of high avian endemism, for example.

Wang et al. (2010) reviewed the sensor types and applications for ecology, biodiversity and conservation. High spatial resolution provides the benefit of accuracy in identification of small objects previously only obtained by aircraft, and analysts can employ high spatial resolution imagery to assess the accuracy of moderate or coarse resolution imagery. With the use of high spatial resolution data, we have the ability to quantify canopy cover and spatial structure of critical habitats, offer essential baseline information for biodiversity monitoring and management, and map changes of heterogeneity of habitats. Hyperspectral data provide the best way to discriminate fine-scale, species-specific landcover and plant properties. Moreover, we can compare the spectral signatures collected to existing spectral libraries and use the information to classify, characterize, and document changes in landcover. Thermal sensors detect emitted energy. The gathered information informs our understanding of the land-energy balance and the relationship between thermodynamics and the principles of ecological patterns of structure and function. With thermal sensors, we can study disturbances such as fire and measure evapotranspiration and soil moisture. Light and detection and ranging (LiDAR) provides details on forest structure, and we can measure canopy height, biomass, and volume for critical habitat investigations.

Advances in sensor technology and algorithms contribute to the further development of RS for ecological study and biodiversity conservation (Wang et al. 2010; Pettorelli and O'Brien

2014). For instance, Andrew et al. (2014) developed a framework to show how land use and landcover derived from RS data could provide estimates of ecosystem services and processes relevant to biodiversity conservation. But poor collaborations between landscape ecologists and remote-sensing specialists have impeded the rapid development of RS approaches as compared to the quantitative spatial-analytical approaches in conservation management (Mairota et al. 2015). Few ecologists have expert knowledge in RS technologies and advanced data analyses, and few remote sensing scientists have expert ecological knowledge (Pettorelli and O'Brien 2014). Strengthening collaborations will help achieve the full potential of RS to support wildlife and resource management (Pettorelli and O'Brien 2014; Buchanan et al. 2015).

Research Goals and Significance of the Dissertation Research

Conservation requires adequate methodologies for rapid assessments, monitoring, and geospatial tools and spatial analyses to access management options (Brown 1999). However, since the geospatial technologies, modeling, and dissemination tools may not be included many conservation scientists' and decision-makers' basic training, those methodologies must be more readily available and widely *useful* and *usable* (Busby 2002; Bregt et al. 2002; Buchanan et al. 2015). Instead of concentrating on the development of methods and products, GIS and RS specialists could attempt to understand and meet the needs of the conservation community (Buchanan et al. 2015; Palumbo et al. 2016).

This research was undertaken with this in mind—to benefit the conservation community, either through the delivery of accessible biogeographic methods or information to further the restoration or maintenance of biodiversity. Prevention of biodiversity decline is the best way to conserve biodiversity, and by preserving ecosystems, we preserve the services upon which we depend. With the habitat analyses described in the following chapters, we can more *quickly*

identify protected areas or threats to biodiversity. This is extremely important considering the accelerated rates of biodiversity loss and extinction.

As a primary goal, this dissertation endeavored to fill the research gaps and offer the conservation community more widely useful and usable GIS and RS models for biodiversity conservation. Secondary goals of the research were (1) to contribute to specific conservation programs for critically endangered species, (2) to inform about the status of habitat, and (3) to develop simple but effective methodologies for top conservation research priorities. The research focuses on the function within the population-species level and in the trends in habitat of a target species. While not a specific objective, the research outcomes may influence public policy.

Format of the Dissertation

In conformance with University of Cincinnati's Department of Geography requirements, this dissertation follows a three-article format with the over-arching theme of the application of geospatial tools for biodiversity conservation.

The second chapter presents the first paper entitled "Putting the wild back into wilderness: GIS habitat suitability modeling for extirpated species." Restoration of an extirpated species, especially a keystone species, is one of the primary means of conserving biodiversity (Fielder and Groom 2006). In this paper, a GIS habitat suitability model based on ecological niche theory incorporated a threat surface layer to determine suitable and unsuitable habitat for the potential reintroduction of the rare and critically endangered red wolf (*Canis rufus*) in historic ranges within the Daniel Boone National Forest in Kentucky. Road density is the delimiting factor for wolves. Previous wolf habitat suitability models assumed equal risk for the roads; however, roads with higher traffic volume and speed pose greater mortality risks for

species. Therefore, this habitat suitability model ranked the roads by mortality risk using the kernel density function in ArcMap and merged the road density layer with the reclassified land cover data to identify potential restoration sites based on known wolf habitat criteria.

In the third chapter, the second paper introduces a hybrid change detection methodology. The aim of "An approach for rapid change detection of semi-arid riparian habitat using color-infrared aerial photography for habitat assessment of the southwestern willow flycatcher: A case study in Mesquite, Nevada" was to reduce the time involvement and complexity inherent in traditional change detection methods. Misclassification with RS data results in low accuracy in change detection (Bregt et al. 2002), and the heterogeneity in aerial photography often precludes automated classification. Rather than co-register and classify the multi-date aerial photography, the study relied on the SPEARS Vegetation Delineation tool in ENVI software to delineate the riparian vegetation with NDVI analysis and to overlay the results from two different years in ArcMap. This enabled the visualization and quantification of habitat changes of the critically endangered riparian-obligate songbird. *The Journal for Conservation Planning* has accepted this paper for publication in Fall 2017.

The fourth chapter imparts the third paper, "Multi-Sensor Change Detection: An Introduction to an Integrated Remote Sensing and Geographic Information Systems Approach Using High-Resolution Data." This research adopts the methodology, target species, and study area of the previous paper to ameliorate the problem of multi-sensor imagery data. With the advent of new sensor platforms and technologies and the difficulty obtaining same sensor data for a study area, a need for change detection methods to handle multi-sensor imagery data has arisen. Computationally complex, for the most part, recently developed multi-sensor change

detection techniques may exist beyond the grasp of non-RS experts. This paper delivers an easier way to monitor changes in the quality and extent of habitat for the conservation community.

The fifth and final chapter briefly summarizes the findings and connect these findings to the larger context of conservation matters. The research implications for practice, limitations, and future directions are included as well.

CHAPTER TWO

Putting the wild back into wilderness:

GIS habitat suitability modeling for extirpated species

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Abstract

Reintroduction of a species is an important conservation strategy, and successful restoration depends on following recommended guidelines. One vital step involves determining habitat quality and suitability for reintroduction of extirpated species. In this pilot study, we explored a deductive approach with the use of kernel density estimation for habitat suitability modeling for the potential reintroduction of the endangered red wolf (*Canis rufus*) to the Daniel Boone National Forest in Kentucky. In previous research on wolf habitat suitability, the logistic regression model used the simple density function to calculate road density, the delimiting factor for wolf populations; however, the model failed to accurately predict wolf recolonization. Roads with higher traffic volumes and areas with greater road densities should pose greater risks to wolf mortality, and simple road density may not be an adequate measure to such purposes. This research, therefore, ranked roads by mortality risk and demonstrated the efficacy of kernel density estimation in Geographic Information Systems as a means to weight the road density and to predict suitable wolf habitat. While viewed as the most reliable contouring method in ecology, kernel density estimation has only been applied to home-range analysis, animal movements and resource use, measurements of overlap areas of species distribution, and other properties of the location such as soil, temperature and photosynthetic rate. Road analysis with kernel density estimation has been limited to networks in the urban environment, such as traffic monitoring, accidents, and bus stops. This method, though, may provide a better picture of the spatial reality of road influence on the likelihood of wolf persistence in a habitat. When applied to this system, the deductive habitat model using kernel density estimation and the mortality-risk ranked roads resulted in predicting nine potential reintroduction sites; whereas, the habitat model with road density calculated with the simple density function predicted no potentially suitable sites.

Keywords: Geographic information systems; habitat suitability modeling; kernel density estimation; mortality-risk ranks; rare species

Introduction

To mitigate biodiversity losses and extinction threats, conservation efforts play a critical role, and one important strategy frequently employed is the reintroduction of a species, presently extirpated or extinct in the wild, into its historic habitat and range (IUCN, 1998). Effective reintroduction must begin with the vital step to ensure habitat quality and suitability for the species (Kleiman, Price, and Beck, 1994; Griffith et al., 1989; IUCN, 1998; Cheyne, 2006) and to identify high quality habitat that will enhance the species' fitness ((Mitchell and Powell, 2002; Chandler and King, 2011). The suitable site should possess the basic habitat requirements for survival and sustainability of a viable population (IUCN, 1989; Cheyne, 2006). In general, only a small number of habitat features, such as abundant prey and proximity to water, are considered important in determining species fitness (Yapp, 1922; Duerksen et al., 1997). These limited key variables serve modeling well by simplifying the complexities of species-habitat relationships to manageable components in the analysis.

The ecological niche theory underpins habitat suitability modeling. As a basic ecological concept, ecological niche describes the position of an organism within an ecosystem, combining the life-sustaining ecological conditions with the species' functional role (Hutchinson, 1957; Polechová and Storch, 2008). Since the ecological niche theory links fitness to the environment (Hirzel and Le Lay, 2008), an application of the ecological niche in the habitat suitability model (HSM) provides the environmental variables upon which to base the probability of species occurrence within an area. Therefore, to ascertain validity and reliability of habitat suitability modeling, it is paramount to understand the ecological niche of a species.

The meaningful parameters have to be derived and researchers may adopt one of two approaches: the inductive or the deductive approach. The inductive approach is based on

empirical data and correlates observations of species' occurrence with the biophysical properties of the locations in which they are found. However, the lack of available data and an understanding of the complex species-habitat relationships in some cases constrains the inductive approach in some cases. Hence, when modeling rare species, common species that are rarely studied, or species with low detectability, the deductive approach is more reliable and appropriate (Ottaviani, Giovanna, and Boitani, 2004). The deductive approach defines habitat criteria based on the ecological niche theory and *a priori* expert knowledge of the habitat requirements (not observations) and generates predictions of suitable habitat.

This paper reports a pilot study in habitat suitability modeling (hereafter referred to as modeling) for the potential reintroduction of a critically endangered species, the red wolf (*Canis rufus,* Audobon and Bachman, 1851). In 1980, *C. rufus* was removed from the last remaining habitat in their historical range from Pennsylvania to Florida and as far west as Texas, declared extinct in the wild, and placed in a recovery program that entailed captive breeding and reintroduction into the wild. By 1987, the United States Fish and Wildlife Service (USFWS) had reintroduced *C. rufus* to the Alligator River National Wildlife Refuge (ARNWR). Surveys performed in 2006 indicated that the population of *C. rufus* rose to 208 in captivity and 130 in the wild (USFWS, 2007). However by 2013, only 50-75 individuals roamed in five North Carolina Counties due to illegal hunting (USFWS, 2016). Analysis suggests a population of 550 (330 in captivity, 220 in the wild) would be stable for genetic diversity (Phillips, Henry, and Kelly, 2003; DeBelieu, 1991). With this goal in mind, the USFWS called for additional establishment sites for the rare and endangered *C. rufus* (USFWS, 2003). It is now imperative to locate other reintroduction sites for red wolf persistence. This research may contribute to the Red

Wolf Species Survival Plan by identifying and evaluating potentially suitable habitat for future reintroduction.

Many HSMs require presence-absence or presence-only data to predict species distribution and occurrence. However, the absence of *C. rufus* throughout its historic range precludes the use of these statistical methods, and the lack of data presents a challenge for modeling. This research, therefore, implemented the deductive approach in geographic information systems (GIS) and the use of kernel density estimation (KDE) on mortality-ranked roads to model habitat suitability for *C. rufus*.

While viewed as the most reliable contouring method in ecology (Hemson et al., 2005), KDE has only been applied to home-range analysis (e.g., Worton, 1989), animal movements and resource use (e.g., Hooge, Eichenlaub, and Solomon, 2000; Marzluff et al., 2004; Fortin et al., 2005), measurements of overlap areas of species distribution (e.g., Fortin et al., 2005; Ridout and Linkie, 2009), and other properties of the location such as soil, temperature and photosynthetic rate (Seaman and Powell, 1996). To date, road analysis with KDE has been limited to networks in the urban environment, such as traffic monitoring (e.g., Yoon et al., 2007), accidents (e.g., Anderson, 2006), and bus stops (e.g., Robinson, 2008). Analyzing road density for habitat suitability modeling is an innovative use of the KDE approach.

The objective of this paper is to demonstrate the appropriate use of this unique approach for modeling in GIS. The specific aim is to develop a simplified, yet widely applicable model that captures the essence of the ecological niche as best known from the literature and produces a reliable predictive map of habitat suitability without the need for information on actual species' occurrence.

Materials and Methods

Study area

With 8498 km² of mixed-mesophytic forest, of which the National Forest Service manages 2823 km², the Daniel Boone National Forest (DBNF) in the Appalachian foothills of Eastern Kentucky is one of several remaining national forests in the historic range of *C. rufus* with sufficient wildlands to support viable populations. (Fig. 1)

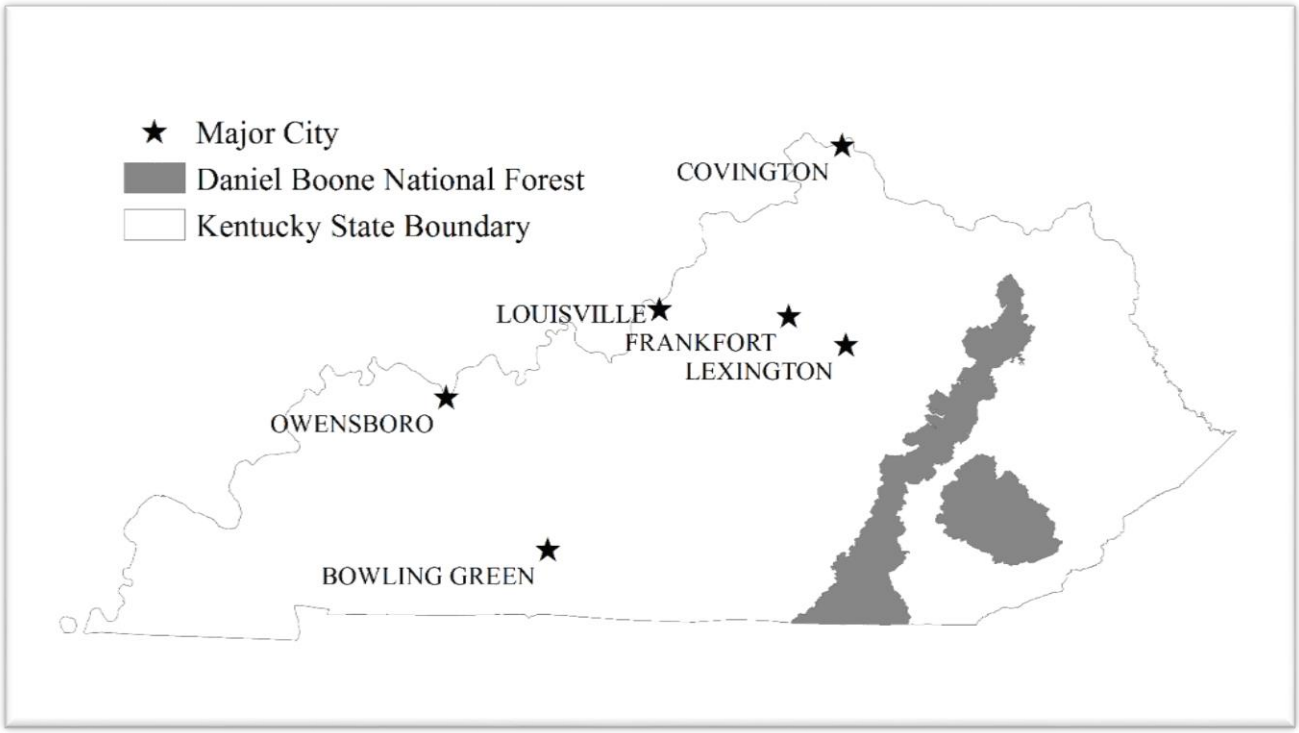

Figure 1: Study area map: Daniel Boone National Forest, Kentucky.

Other sizable national forests include the Ozark National Forest in Arkansas, Shenandoah National Park in Virginia, Nantahala National Forest in North Carolina, and the Great Smoky Mountains National Park on the border of Tennessee and North Carolina, where the second reintroduction attempt failed because of the high pup mortality (USFWS, 1998). All of these potential sites warrant evaluation, but the success of recent elk (*Cervus elaphus,* Linnaeus, 1758)

reintroduction to the DBNF influenced the study area selection. The habitat conditions for the elk restoration zones—low human population, long distance from row crops and urban centers, and heterogeneous landscape of forest, grassland, and shrubland (Larkin et al. 2004; Maehr, Grimes and Larkin, 1999)—suit *C. rufus* as well. Although not known, elk fawn may supplement the diet of *C. rufus* during the calving season. The oak-hickory (*Quercus-Carya*) and oak-pine (*Quercus-Pinus*) forests, in addition, support abundant prey species, including the white-tailed deer (*Odocoileus virginianus*), raccoon (*Procyon Lotor*), and marsh rabbit (*Sylvilagus palustris*) that constituted 88.7% of biomass consumed by *C. rufus* in Alligator River National Wildlife Refuge (Phillips, 1995).

Habitat and model criteria

Wolves behave as habitat generalists, able to persist in any area with sufficient prey and shelter and minimal habitat fragmentation, disturbance, and harassment by humans (USFWS, 2007). In North Carolina, the restored *C. rufus* have occupied a mosaic of landscapes—wetlands, pine forests, upland shrubs, croplands, and pocosins and have utilized edge interfaces for travel and prey access (USFWS, 2007).

As a model index, Mech (1995) denoted road density as the "yardstick" by which agencies and recovery teams measure wolf habitat suitability. Wolves do not have an aversion to roads and travel roadways with lower traffic volumes (Mladenoff et al., 2009; Wydeven at al., 2006). Only roads with moderate to heavy traffic pose problems for wolves due to increased risk for wolf mortality from vehicular accidents (Mladenoff and Sickley, 1998). In North Carolina, during the years 1987-1994, motor vehicles caused 30% of the deaths of *C. rufus* (Phillips et al., 2003). Highways and major roads with frequent traffic, not only heighten risks, but form significant barriers to wildlife movement within the forest, and subsequent fragmentation creates

potentially small patchy habitats (Heilman et al., 2002). Road density approximates human activity and the potential for human-caused mortality and has the greatest explanatory effect on the wolf-habitat relationship (Harrison and Chapin, 1998; Mladenoff et al., 2009). Highest natural mortality for wolves occurred in habitats with road density values between 0.63 and 0.84 km/km² and highest human-induced mortality occurred in habitats with road densities between 0.84 and 1.14 km/km² (Wydeven at al., 2006).

Mladenoff et al. (1995) utilized GIS to analyze road densities within 14 wolf-pack territories in Wisconsin, finding highly suitable habitat with road densities less than 0.45 km/km² (mean 0.23 km/km²). Fuller et al. (1992) determined road densities less than 0.7 km/km² in wolf pack areas in Minnesota. Other researchers (Corsi et al., 1999; Frair, 1999; Harrison and Chapin, 1998; Houts, 1999; Kohn et al., 2000; Mladenoff and Sickley, 1998; Mladenoff et al., 1997; Ratti et al., 1999; Shelley and Anderson, 1995; Unger, 1999) have since incorporated road density in habitat studies and determined it the best predictor for suitable wolf habitat.

Mladenoff et al. (1995) developed the standard for wolf HSM based on road density using logistic regression. Researchers have since applied this approach for wolf-habitat predictions (Brito et al., 1999; Glenz et al., 2001; Houts, 2003; Keating and Cherry, 2004; Mladenoff et al., 1995; Mladenoff and Sickley, 1998; Mladenoff et al., 1999; Ratti et al., 1999). While popular, these HSMs follow an inductive approach and are usually built with classical statistic methods (e.g., logistic regression) that require a wealth of presence and absence data. Considering this data limitation for *C. rufus*, the logistic regression model is not feasible for *C. rufus* modeling, and the HSM for its reintroduction necessitates the deductive approach.

Modeling in GIS

The deductive approach in GIS produces a predictive map of habitat suitability. By representing the potential suitable habitat for reintroduction in a visual, rather than textual or tabular, format, the predictive map can expedite the interpretation of the statistical and thematic analysis, which may better guide further decisions (Convis, 2001). GIS has emerged as a prominent, time- and cost-effective tool in conservation science (Convis, 2001; Bishop et al., 2002).

Pursuant to the deductive approach, model development began with the selection of key variables drawn from the literature. Two factors were determined to be the best predictors for habitat suitability for *C. rufus*: road density, as the delimiting factor for survival, and landscape composition, which captures the habitat preference of *C. rufus.*

To conduct the GIS analysis, the forest boundary layer, the road data layer, and the landcover classification layer were acquired from Bill Luhn, the GIS Coordinator for the DBNF District Office. The vector road data from the DBNF District Office included all roads, paved and unpaved, public and private, within the DBNF. In total, 3691 line segments existed, representing the highway, arterial, collector, and local roads. They were classified by road type: undefined highway, free flowing mixed traffic, congested during heavy traffic, flow interrupted/limited use, and slow flow or blocked. The latter two types of roads consisted of forestry, fire service, and closed secondary roads and were removed from the analysis due to their assumed negligible risk to wolves.

In earlier models, road types assumed equal weights in the road density. Ratti et al. (1999) performed the only feasibility study where road density was determined separately for four different road classes, but road density for each class was calculated by dividing the length of road by the area. No weights for mortality risks associated with road types were included.

However, road kills of black bear (*Ursus americanus,* Pallas, 1780) for example, occur primarily on paved roads with heavy, fast-moving traffic (Rogers and Allen, 1987). The number of road kills depended upon the density of paved roads and the amount of traffic. Higher volumes of traffic with faster moving-vehicles increase the chances of faunal mortality, and this ecological effect of road kills associated with traffic intensity has been well-documented (e.g., Jackson, 2000; Kobylarz, 2003; Maine Interagency Work Group on Wildlife/Motor Vehicle Collisions, 2001; Noss et al., 1996; Seiler, 2001; Seiler and Helldin, 2006). Vehicular collisions with wildlife can devastate populations of geographically isolated or rare species (Forman et al., 1997). For a rare predatory species, such as *C. rufus,* road types with high traffic volume and high traffic speeds will therefore have the greatest impact on species fitness. Incorporating this mortality-risk assumption into the model, we created a new field in the road attribute table and ranked the roads by traffic volume and associated mortality-risk, where asphalt highways posed the greatest threat to wolf mortality and local gravel roads the least.

Further contrast to earlier models, the road density in this research was calculated as a weighted probability function with KDE rather than a simple density function (SDF). Simple density is calculated by adding the lines that fall within an area and dividing the sum by the area size; whereas kernel density calculations sum all the values of the kernel surfaces—the smoothed curved radius around a line, with density greatest on the line and diminishing outward—and produce smoother results than histograms. The width or variance of the kernel affects the amount of smoothing (Seaman et al., 1998). If under-smoothed, the curve will appear spiked with spurious peaks; while a larger bandwidth will smooth away the spurious features, over-smoothing will also smooth away relevant features (Wand et al., 1991).

In GIS spatial analysis, the KDE is calculated with the Epanechnikov quadratic K estimator, an optimal smoothing function determined by $K(t) = \frac{3}{4}(1-t^2)$, where $|t| = d/h \leq 1$ (d = the distance between the cell and the line in the dataset). Unlike other interpolation techniques such as kriging-cokriging, trend surface, and regressions, KDE aims to produce smooth commutative density functions (Amatulli et al., 2007) and hot spots. The density estimate will be high in areas with many observations and, conversely, low in areas with few observations (Seaman and Powell, 1996). KDE generates these hotspots quickly from large datasets and provides a statistical, visual outcome of a more realistic continuous model of distribution patterns than other hotspot or clustering techniques (Anderson, 2006).

The kernel process is a probability density function (k) placed over a data point or line, and the addition of *n* components constructs the estimator (Worton, 1989). The kernel defines the shape of the weighting function and represents a density. Therefore, the estimation reflects a "true probability density function." The smoothing parameter controls the variation in each component, with direct correlation existing between size of the bandwidth h and the scope and scale of detail in the data observations. The probability function is expressed as

$$\widehat{f}_h(x) = \frac{1}{Nh}\sum_{i=1}^{N} K\left(\frac{x - x_i}{h}\right) \quad \textbf{(1)}$$

where *K* represents the kernel and *h* represents the bandwidth. The bandwidth defines the radius of the circle of each grid cell, and, in GIS, the default bandwidth measure is based on the geographic extent of the point or line patterns. While the selection of the bandwidth is important, the process of selection is more art than science.

KDE, as a non-parametric estimation, has the flexibility to accurately estimate non-convex, multimodal, and irregular shaped distributions (Seaman et al., 1998). It also relies on fewer

assumptions, which may result in fewer misinterpretations. Since this research ranks the roads by mortality risk, the use of the non-parametric KDE is more appropriate.

To model the second key variable, the landscape composition, we followed the guidelines of the revised logistic model to exclude agricultural lands (Mladenoff et al., 2009). While the Milltail pack occupies territory on the crop lands in ARNWR, subsisting off smaller prey items such as rodents and lagomorphs (Phillips et al., 2003), the farms have no human occupants. Historically, the majority of wolf-human conflicts stemmed from agricultural and pastoral practices, whereby wolf depredation of domesticated livestock instigated eradication measures (DeBlieu, 1991; Mech, 1995; Musiani and Paquet, 2004). The assumption that wolves do not belong in humanized landscapes may be unjustified (Lynn, 2002), but until human attitudes and behaviors change toward predators, humanized landscapes will be treated as unsuitable. We therefore reclassified the landcover classification layer according to wolf suitability, assigning areas with crop and pasture as null (0) for unsuitability, and the remaining forest, water and wetland classes as positive (1) for suitability.

The final steps of the modeling process involved merging the two raster datasets to produce the suitability map. With a low probability of wolf persistence in areas with road density greater than 0.68 km/km² or 0.7 km/km² (Thiel, 1985; Fuller et al., 1992; Wydeven et al., 2006), we used this threshold in the raster calculation. The output then delineated a map of unsuitable ("high") road density and suitable ("low") road density. With the raster calculator, areas with suitable road density and forested, non-agricultural landscapes were merged and determined. We then converted the *C. rufus* HSM from raster to vector in order to highlight habitat patches with a minimum area of 50km² as potential restoration sites.

To assess the efficacy of the kernel method, we recalculated road density with SDF based on the shape lengths and created an alternative *C. rufus* HSM to compare. All other steps remained constant.

Results

Modeling in GIS with the KDE approach resulted in the identification of nine patch areas with restoration potential for *C. rufus* (Figure 2). In contrast, the HSM built with SDF predicted no suitable sites for reintroduction (Figure 3).

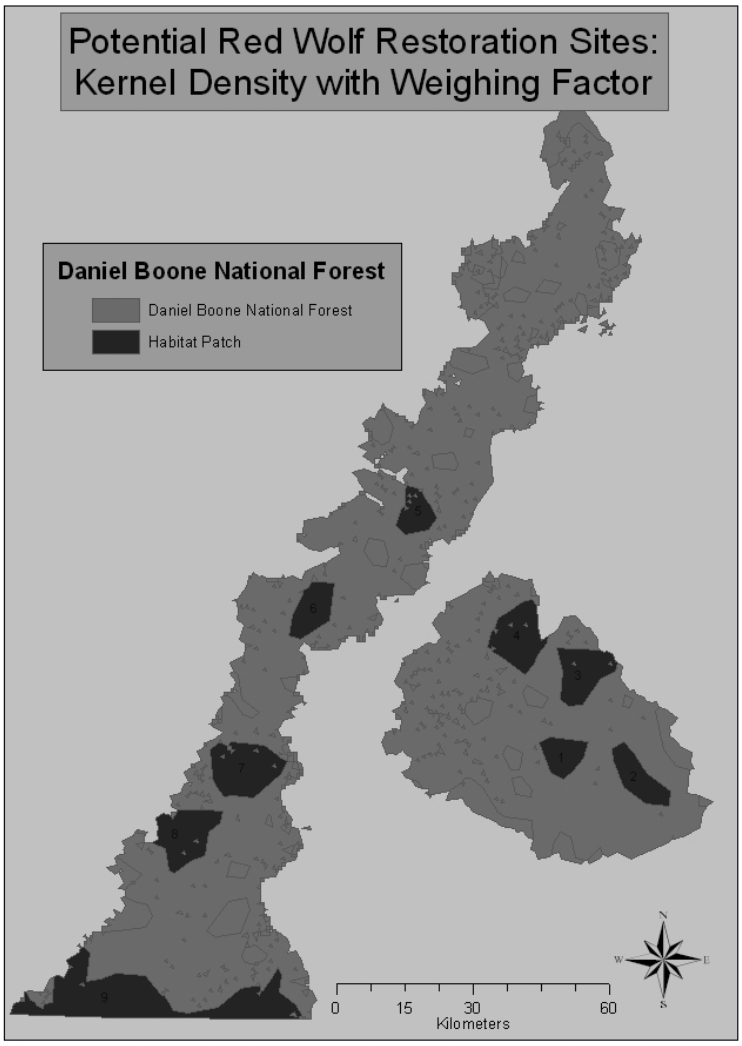

Figure 2: Map of the nine potential red wolf restoration sites as predicted using KDE and the mortality-risk weighing factor. Only patches with areas greater than 50 km², low road density and suitable habitat were selected as "optimal" sites for restoration.

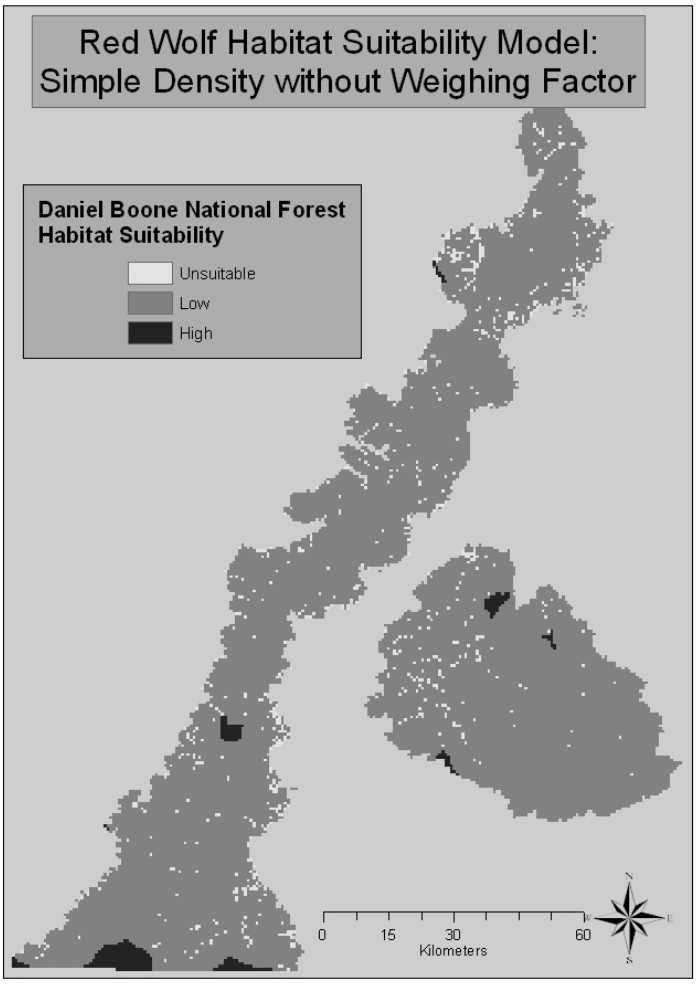

Figure 3: Red wolf suitability map based on simple density without mortality-risk weighing factor showing unsuitable habitat, low suitability habitat, and high suitability habitat. This method returned no habitat patches greater than 50 km².

In total, the kernel density model predicted 1207 km² of suitable habitat for reintroduction purposes. Home ranges may extend anywhere between 25 to 130 km²; thus, the habitat patches within DBNF could theoretically, based only on size, support 9.3 to 48.2 packs, or up to approximately 241.4 individuals. Out of the nine potential sites identified by the model,

two habitat patches in the Red Bird District might possess more optimal habitat because of their forest buffers, connectivity to other patches, and lack of agriculture. The extent of pasture and crop land in the forest- dominated landscape of the DBNF appears more concentrated along the southeastern edge of Stearns District, the Morehead District, and the western and northwestern region of the Red Bird District. Unsuitable areas with high road densities, on the other hand, are associated with Interstate 75, major state byways, the Red River Gorge geological area, and the large lake recreation areas in the north (Cave Run Lake) and south (Cumberland Lake).

The kernel of the matter: road type

The results of the modeling presented in this paper predicted the probability of high-quality, suitable habitat and the occurrence of *C. rufus* in the DBNF; yet, the HSM is only to be viewed as a hypothesis of the species-habitat relationships to be tested (Mitchell and Powell, 2002; WSC, 2006). As the key delimiting factor affecting the fitness of *C. rufus*, road density is thought to exert the most control on the habitat selection (Corsi et al., 1999; Frair, 1999; Harrison and Chapin, 1998; Houts, 1999; Kohn et al., 2000; Mladenoff et al., 1995, 1997, 2009; Mladenoff and Sickley, 1998; Mech, 1995; Ratti et al., 1999; Shaffer, 2007; Unger, 1999; Wydeven et al., 2006). In this pilot study, we utilized KDE and a weighting factor to elucidate on the relationship between road density and habitat suitability and found the type of road made a difference in the inquiry. Frequency and speed of travel is higher on asphalted highways and state byways than on gravel country roads, and, therefore, the chances of vehicular strikes with wildlife will increase on roads more often travelled (Jackson, 2000; Kobylarz, 2003; Maine Interagency Work Group on Wildlife/Motor Vehicle Collisions, 2001; Noss et al., 1996; Rogers and Allen, 1987; Seiler, 2001; Seiler and Helldin, 2006).

As a means to validate the model without presence/absence data, vehicle-deer collision data served as a surrogate indicator for road risks (Figure 4, Kentucky State Police). Webb (2012) reviewed the factors associated with deer-vehicle collisions and surmised the highest risk roads for animal-vehicle collisions have higher speed limits, traffic volume, load densities, and were proximal to areas with high forest cover. These type roads were also considered the highest risk for wolf mortality as well. Ranking the roads by mortality risk is a significant step in this model and deviates from previous models, and by using vehicle-deer collision data, we demonstrate how well the model with the weighting factor performed. As a result, this model did not predict any suitable habitat in Rowan County located in the northern Morehead District, where the highest deer-car collisions occurred. On the contrary, the model predicted four potential restoration sites in Leslie and Clay Counties located in the Red Bird District, where very low deer-car collisions occurred. It would seem the assumption built into the methodology is realistic. In the model, the highways and state routes received the highest mortality-risk rank, and, in reality, the higher deer-car collisions occurred on those road types with high speeds and higher traffic volumes in the northern and southern regions of the DBNF, areas associated with State Route 60, Interstate 64, and Interstate 75. Traffic data, although unavailable for the study region, would facilitate further model validation.

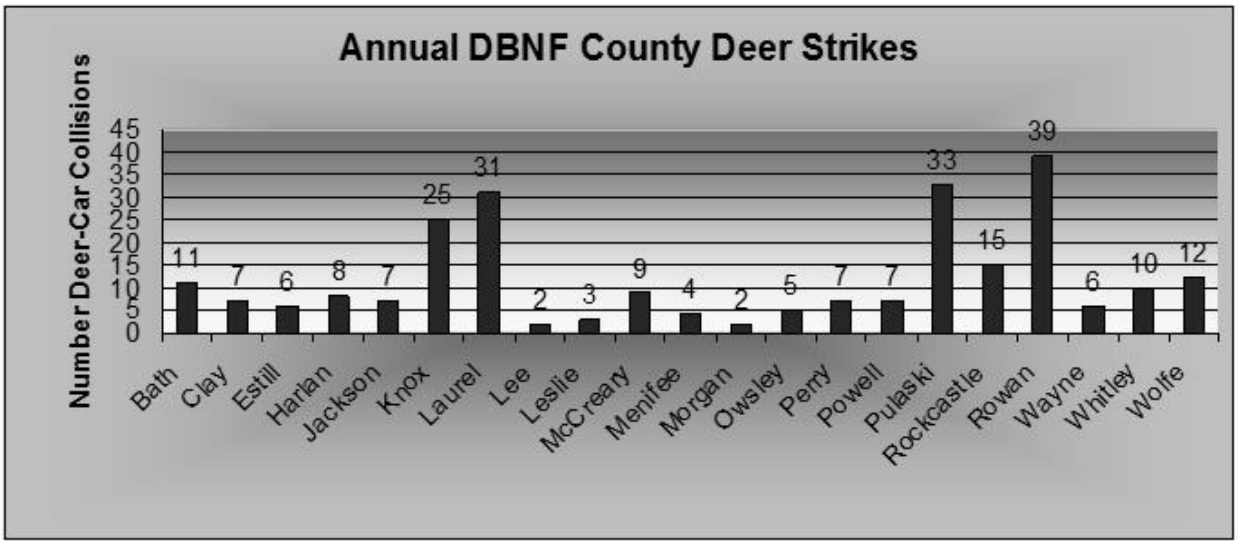

Figure 4: Bar graph depicting the number of deer-vehicular collisions. Data source: Kentucky State Police **http://www.kentuckystatepolice.ky.gov/deerauto.htm**

Integrating the severity of threat may have improved the model's performance in portraying the spatial reality of road density on the availability of suitable habitat. The relationship between road influence and wolf-habitat is explained by mortality risk as expressed in the weighting factor. Without the ranking of mortality risk, the road type made little difference in the modeling efforts, and, as a result, high road density occurred in areas with roads carrying an assumed lower mortality risk. The logistic regression models (Mladenoff et al., 1995, 2009) may have performed more accurately if the roads were treated differentially and assigned a mortality-risk rank by road type. Still, an interesting future study would be to attempt to model the distribution of *C. lupus* with this pilot study's methodology to ascertain its validity. Once validated, the model should then be applied to other Eastern forests with sufficient wilderness to locate additional restoration sites.

Conclusions

Although a rare and endangered species can be raised in captivity, its ultimate survival is dependent on its restoration to the wild (Clark and Westrum, 1989). Successful reintroductions that enhance long-term survival of a species, re-establish a keystone species, increase or maintain biodiversity, or provide long-term economic benefits to local population (IUCN, 1998), in turn, depend upon effective modeling. This research introduced a new HSM methodology for wolves, but it is applicable to any species, especially for those rare and endangered species, species with low detectability, and species uncommonly studied, where presence data are inadequate or missing and where density of roads, breeding sites, prey abundance, environmental or biological threats, etc. is the important variable.

KDE affords advantages to modeling in GIS, providing more realistic continuous density surface models and easily, visually identifiable hot spots (Anderson, 2006). It better handles dynamic data features and helps in understanding geographic pattern changes (Anderson, 2006). Using spatially explicit information on species' fitness and spatial statistics, this HSM is a vital step toward a conservation aim to protect against biodiversity loss and restore a keystone species to its historic range.

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CHAPTER THREE

An approach for rapid change detection of semi-arid riparian habitat using color-infrared aerial photography for habitat assessment of the southwestern willow flycatcher: A case study in Mesquite, Nevada

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ABSTRACT

Highly productive, diverse, and ecologically important riparian ecosystems in the southwestern United States have declined 90% since historic times. Since at least 60% of all vertebrate species and 70% of the rare, threatened, and endangered species, such as the southwestern willow flycatcher (*Empidonax traillii extimus*), depend on these riparian zones in semi-arid environments, it is imperative to document, quantify, and characterize the changes in these habitats. This information will be instrumental for managers, planners, and decision-makers to further understand the potential environmental and biological vulnerability of the area and take action to reduce the vulnerability and to conserve or restore these important riparian ecosystems. To achieve this aim, we introduce an original change detection methodology using color-infrared aerial photography, ENVI's SPEAR Vegetation Delineation Tool, and geographic information systems. This technique significantly reduces processing time and enables researchers not only to produce maps highlighting areas of change, but also to quantify those changes. The results of our pilot study on a small, isolated southwestern willow flycatcher breeding site in Mesquite, Nevada—using the aforementioned technique—showed a 42% decrease in vegetation cover from 2004 to 2010. The approach proved to be sufficiently accurate and easy to implement, a distinct advantage over other change detection techniques.

Keywords: Change detection, remote sensing, SPEAR Vegetation Delineation Tool, Geographic Information Systems, riparian ecosystems, semi-arid environments, southwestern willow flycatcher, conservation

INTRODUCTION

Southwestern U.S. riparian ecosystems rank amongst the rarest in the Western Hemisphere (Krueper 1996) and the most diverse, complex, productive, sensitive, and fragile in North America (Johnson et al. 1977; Johnson 1978; Knopf and Samson 1994; Kondolf et al. 1996; Nilsson and Svedmark 2002; and others). Constituting only 1-3% of the southwestern landscape, these narrow, visually distinct riparian zones along ephemeral, intermittent, and perennial streams contain disproportionately rich vegetation as compared to surrounding steppe, shrub, and desert areas (Nilsson and Svedmark 2002). Wilson (1979) referred to the ecological importance of riparian vegetation as "the aorta of an ecosystem". The structural complexity of multiple canopies within riparian habitats creates multiple niches, and the availability of moisture and cool, shady microhabitats in otherwise dry, hot regions yield high species diversity (Kondolf et al. 1996; Patten 1998). For 75% of the local wildlife species, these riparian habitats are essential during at least one phase of life (Kondolf et al. 1996). In the southwest, 60% of all vertebrate species and 70% of rare, threatened, and endangered species depend on these riparian zones (Fischer et al. 2001; Poff et al. 2012). They support a higher breeding diversity of birds than all other western habitats combined as well as the highest noncolonial avian breeding densities in North America (Johnson et al. 1977), providing critical habitat for more than 50% of the songbirds (Knopf and Samson 1994; Hatten et al. 2010), including the endangered southwestern willow flycatcher (*Empidonax traillii extimus*) (hereafter referred to as SWF). Like the proverbial canary in the coal mine, the SWF acts a prime indicator of ecosystem health of southwestern riparian zones (McCarter 1996). Its status as the most endangered riparian obligate in the Southwest (Suckling et al. 1992) signals significant changes in the riparian ecosystems.

The diversity of environments and vegetation communities in southwestern riparian ecosystems manifests in a wide range of valuable functions and services, including water filtration, bank stabilization, nutrient cycling, sediment load reduction, scenic beauty, natural resources, and recreational opportunities (Patten 1998; Poff et al 2012). Yet, despite their extreme ecological, economic, and societal importance, these valuable ecosystems have declined 90% since historic times (Ellis et al. 2009; Hatten et al. 2010). The greatest impact of riparian zone destruction has occurred in the Southwestern and Southeastern United States (Fischer et al. 2001). Dams, water diversions, and groundwater withdrawal have altered or eliminated the natural hydrology that enable the formation of Southwestern riparian forests (Marshall and Stoleson 2000; Paradzick and Hatten 2004). Other human activities, such as urbanization, land development, mining, agriculture, and livestock grazing, further compound the negative impacts, either with direct or indirect effects (Patten 1998).

As a result, the remaining riparian habitat patches are smaller, more isolated, and more susceptible to degradation by stochastic events, and endangered species populations, such as the SWF, are more vulnerable to local extinction (Ellis et al. 2009). The SWF survival, specifically in breeding success, depends on the distribution and abundance of breeding habitat types within large, active floodplains with young, wide and dense stands of riparian vegetation (Sogge et al. 1997; Hatten and Paradzick 2003; Durst et al. 2008; Paxton et al. 2007), consisting of willow (*Salix* sp.), seep-willow (*Baccharis* sp.), tamarisk (*Tamarix* sp.) arrowweed (*Pluchea* sp.), or Russian olive (*Eleagnus* sp.) and situated near lentic water (Finch 1999; Sogge and Marshall 2000; Graf et al. 2002). Habitat loss, degradation, and modification, however, threaten the SWF (Sogge et al. 1997; Stoleson et al. 2000), forcing the subspecies to the brink of extinction. SWF exhibit site fidelity, although not absolute (Unitt 1998), and migrants who search for suitable

patches in shrinking or degrading habitat may face increased mortality risks from competition, starvation, or predation, whereby reducing or eliminating breeding opportunities (Marshall and Stoleson 2000). In turn, the SWF population diminishes. The link existing between the declines in riparian habitat and in SWF populations (Paxton et al. 2007) exemplifies the SWF role as a very good indicator of the health and stability of riparian ecosystems in its home range in southern California, Nevada, Utah, New Mexico, Texas, and Arizona. Therefore, maintaining and restoring riparian environments has become a conservation and management priority in the Southwest (see e.g., Johnson et al. 1977; Knopf and Samson 1994; Finch 1999; Marshall and Stoleson 2000; Stromberg 2001; Hatten et al. 2010). The functional qualities and values, including wildlife habitat, water filtration, bank stabilization, nutrient cycling, sediment load reduction, scenic beauty, natural resources, and recreational opportunities, make them key ecosystems for preserving biodiversity (Nilsson and Svedmark 2002).

This paper, in response, may contribute to the Southwestern Willow Flycatcher Recovery Plan—"the largest and most comprehensive planning and recovery effort for an endangered species" (Graf et al. 2002)—by mapping and quantifying changes in one of the smaller, more isolated habitat patches which are more at risk for local extinction. Documenting, quantifying, and characterizing these changes will alert managers, planners, and decision-makers about potential environmental and biological vulnerability and prompt action to reduce the vulnerability and to conserve or restore the value of riparian ecosystems (Kepner et al 2000).

Since the SWF is an umbrella species, conserving or restoring riparian habitat for the SWF benefits at least 83 other species, including the endangered New Mexico jumping mouse (*Zapus hudsonius luteus*), the yellow billed cuckoo (*Coccyzus americanus*), Chiricahua leopard frog (*Lithobates chiricahuensis*), and the Least Bell's vireo (*Vireo bellii pusillus*). Hence, it is

crucial to be able to quantify and further understand the changes of the SWF habitats for long term management of species and communities.

Geospatial tools, such as remote sensing and Geographic Information Systems (GIS), play vital roles in conservation science. With these tools, ecologists, biologists, land managers, policy-makers, conservationists, nonprofit organizations (NGOs), and even volunteers can harness computer power to analyze, model, and map biophysical data. The first large scale computer modeling for suitable SWF nesting habitat combined a remote sensing and GIS approach to determined predictor variables (Hatten and Paradzick 2003). In their study, Hatten and Paradzick (2003) created riparian-vegetation density grids in four steps: (1) calculation of Normalized Difference Vegetation Index (NDVI) for the relative density and biomass of green vegetation because NDVI strongly correlates with the biophysical property of plants; (2) clustering NDVI into 12 interval-scaled classes; (3) overlay of NDVI classes and satellite imagery to find boundary between riparian and upland vegetation; and (4) clustering riparian forest into 12 interval-scaled density classes. They found NDVI class values ranged from -0.522 to 0.63, whereby riparian vegetation increased with increasing value. The value 0.126 corresponded to the cutoff between riparian and upland vegetation and values greater than 0.336 corresponded to the densest riparian vegetation. Higher NDVI values that indicated higher density of riparian vegetation better predicted SWF breeding activity.

Three studies since have coupled habitat suitability variables and NDVI probability classes to monitor changes in predicted SWF breeding habitat (Paradzick and Hatten 2004; Paxton et al. 2007; Hatten et al. 2010). As a result of modeling, Hatten et al. (2010) determined the NDVI variable that summarized density at the finest scale was the most influential covariate and reaffirmed the importance of dense riparian vegetation as a major selection criterion of the

SWF. Based on these results, NDVI is a key aspect of our change detection approach in this paper, given its importance in SWF habitat suitability modeling.

The visual and statistical approaches of change detection presented in this paper will help achieve the aim to quantify and understand the changes in SWF habitat. Singh (1989) defined remote sensing change detection as "*a process of identifying differences by observing it at different times"*. This process operates on the assumptions that the radiance values reflect the biophysical properties of the land cover and changes in radiance values correspond to changes in land cover (Singh 1989; Coppin and Bauer 1996). Two categories of change can occur: (1) between classes and (2) within classes. A change between classes refers to the conversion of one land cover to a different land cover (e.g., forest to farmland), while a change within classes refers to the modification of the condition of the land cover (e.g., fragmentation or coalescence; expansion, shrinkage, or alteration of shape; or positional shift). Due to the magnitude of global problems arising from land cover/use changes, the applications vary widely and have increased in importance: deforestation; urban sprawl; industrial development; coastal zone changes (e.g., erosion); climate change effects (e.g., sea ice, thaw lakes, glacial mass); farmland loss; crop monitoring; desertification; flooding; soil erosion; plant community changes; wetland changes; forest fires; forest mortality, defoliation, damage estimations; algal blooms; invasive species spread; etc. (*see* e.g., Lu et al. 2004).

While the quantification of change detection most often relies on satellite-derived imagery data (Coppin and Bauer 1996) because of high availability, cost-effectiveness, and broad spatial and temporal coverage, change detection of small, fine-scale riparian ecosystems precludes the use of moderate and coarse resolution imagery (>30 m/pixel) generally acquired from satellites (Fensham and Fairfax 2002; Ihse 2007; Heiskanen et al. 2008; Morgan et al. 2010). the need for high spatial information and higher degree of accuracy necessitates the use of harder to acquire and potentially expensive high resolution imagery (<5 m/pixel); to address

this need, in this research, we attempt to use color-infrared (CIR) aerial photography with 1-meter or less spatial resolution.

Change detection based on CIR aerial photography has most often relied on visual interpretation (Fensham and Fairfax 2002), a highly subjective, laborious method with accuracy of results dependent on the interpreter's skills and difficult to evaluate (Lu et al. 2004; Hyvönen et al. 2011). Aerial photointerpretation involves a person looking at the image to identify elements and objects and interpret their significance. For determining changes, an analyst views multi-date images at the same time, generally by creating maps on clear plastic sheets and overlaying these sheets on a light table or digitizing the imagery and combining the characteristics different datasets into one, searching for differences in identified features. Limits to a person's ability to distinguish small differences in shades of grey and color compound the challenge of manually comparing two images for tonal differences in vegetation.

While the use of aerial photography for monitoring and mapping changes provides an advantage of clear identification and extraction of information, detection of false changes may occur without accurate descriptions and knowledge of plant phenology (Ihse 2007). In an attempt to summarize techniques for manual interpretation and change detection, Ihse (2007) found a high degree of accuracy with CIR aerial photography using color as the main criteria to define vegetation. However, other indicators such as physiognomy, ecological conditions, species, site, topography, substrate, and anthropogenic influences needed combined with color to improve image classification and vegetation mapping used for change detection. This involves a complex system of interpretation. In another summary, Heiskanen et al. (2008) assessed three different visual interpretation methods for the feasibility to monitor changes with aerial photography: complete cover mapping (polygon interpretation), sample plot method, and

transects; however, limitations included subjectivity, difficulty visualizing results, and difficulty in transect selection, respectively. Large variations in some variables emerged between interpreters, and many of those differences were statistically significant (Heiskanen et al. 2008). With any interpretation method, the experience of the interpreter affected the accuracy of the results and magnitude of the change, making an apparent need for highly knowledgeable and skilled analysts. This may restrict who can perform change detection with aerial photography.

More recently researchers have shown interest in CIR aerial photography for change detection beyond the more simplistic overlay analysis and visual interpretation, but these have their challenges as well. Given the limitations of visual interpretation, Everitt and Yang (2010) performed computer supervised classification on CIR aerial photography to quantify giant reed coverage changes with satisfactory results. Their sites, however, contained only four or five plant cover types. Conventional pixel-based image classification, which assumes homogenous features (Johansen et al. 2010), may misclassify pixels for sites, such as the SWF riparian habitat, with high spatial and spectral heterogeneity. The errors in pixel-based classified images often produce a salt-and-pepper appearance. Object-oriented classification may overcome this problem with conventional pixel-based classification (Whiteside and Ahmad 2005) since objects contain several pixels. In a study where Gweon and Zhang (2008) opted for wave-let transform and object-oriented classification of aerial photography, the results showed where the changes occurred but could not identify changes in detail. Still, the advanced segmentation process and mechanisms for classifying objects into classes of contiguous pixels of similar color, texture, shape, context, and tone may seem too cumbersome for non-remote sensing experts.

Selection of the change detection method, therefore, proves challenging for any digital format because the myriad of remote sensor systems, environmental characteristics, image

processing methods, and temporal, spatial, spectral, and radiometric resolutions all impact the success of the change detection analysis (Lu et al. 2004; Hussain et al. 2013). Decisions depend on the objectives of the change detection: to identify locations and types of change; to understand the direction and magnitude of the change; and/or to quantify the changes (Coppin et al. 2004), as well as the knowledge and skill of the analyst.

As the challenges described above demonstrate, remote sensing researchers have an ongoing agenda to develop change detection methods (Hussain et al. 2013), and the need for current information on habitat extent and change in extent is one of the top conservation priorities (Buchanan et al. 2015). However, in general, conservation scientists lack the training and skill to interpret or analyze remote sensing data (Buchanan et al. 2015). A broadly accessible change detection approach would benefit the conservation community.

To meet this need, this study introduces a less complicated and time-effective method for change detection based on CIR aerial photography, Spectral Processing Exploitation and Analysis Resource (SPEAR) Vegetation Delineation Tool in ENVI software (Exelis Visual Information Solutions, Boulder, Colorado), NDVI analysis, and GIS that enables both the visualization and quantification of the vegetation cover losses/gains. Based on trends in SWF habitat and population throughout its range, we expect declines in the area of total vegetation cover over time. Quantification of vegetation cover by aerial photography, though, is limited and historically estimated from subjective assessments using field data (Fensham et al. 2002). Less subjective measures, for example, include a dot-grid technique (*e.g.,* Fensham et al. 2002), a statistical comparison of manual counts of trees within a circle polygon overlay created in GIS (*e.g.,* Plieninger 2006), spectral and factorial analysis (*e.g.*, Couteron 2002), and random-point plot-grid sampling (*e.g.***,** Ucar et al. 2016). We propose a shortcut method to quantify the area of

vegetation cover in GIS without plotting grids or manually counting trees, which, in conjunction with the NDVI analysis in the SPEAR tool, will help achieve the aims of the study stated above. With this in mind, the change detection method described within this paper bridges remote sensing and conservation and helps build capacity.

STUDY AREA

Habitat Requirements

The SWF breed in four general types of riparian habitat: 1) monotypic high elevation willow (*Salix exigua* and *S. geyeriana*) with associated sedges, rushes, and nettles; 2) monotypic exotic, which is characterized by either salt cedar (*Tamarix* spp.) or Russian olive (*Elaenagnus angustifolia*); 3) native broadleaf dominated, often composed of single species Gooding's (*Salix goodingii*) or other willow, or mixed broadleaf trees and shrubs including cottonwood (*Populus* spp.), boxelder (*Acer negundo*), alder (*Alnus* spp.), and Ash (*Fraxinus* spp.); and 4) mixed native/exotic broadleaf trees and shrubs (Sogge et al. 1997). The SWF habitat patches contain a mosaic of dense vegetation interspersed with small openings, open water, and shorter, sparser vegetation, and territories and nests are found near marshy seeps or saturated soil (Sogge et al. 1997). with nest proximity to water, aquatic plants such as cattail (*Typha spp.*), bulrush (*Scirpoides holoschoenus*), and bur-reed (*Sparganium eurycarpum*) are prominent features of SWF habitat (Unitt 1988).

The breeding habitats range from 0.6 hectare to 100 hectares in six states in the U.S. from southern California to southwestern Colorado (Finch 1999); however, nearly half of the total population occupies the smallest territories (Marshall 2000). This distribution within many small sites with small populations and with a small number of sites containing large populations

presents a conservation challenge (Marshall 2000). These small SWF populations with low genetic variability are the most vulnerable to extirpation. Pursuant to theories on habitat fragmentation, persistence of SWF in smaller habitats may depend on the connectivity among the patches and proximity to larger patches (Rocklage and Edelmann 2002). With landscape-level approaches and management at the scale of the drainage basin, habitat conservation thereby plays a crucial role in protecting the existing small, isolated populations and their habitat (Marshall and Stoleson 2000).

The goal of habitat conservation centers on the protection of populations *and* the ecological functions that sustain them. In the SWF habitats, the vegetation functions as substrate for nesting, perching, patrolling, and foraging, cover from predators, and shade to promote the cool, humid microclimate that influences insect prey base and nesting success. The vegetation constituting SWF habitat requires substantial amounts of water. In the Southwest, streams may flow intermittently or perennially, and dams block the natural flooding cycle that scours and replenishes moisture for vegetation growth. Conservation measures, therefore, may depend on the management of water resources to mimic natural hydrological conditions to maintain healthy riparian vegetation and healthy SWF populations.

Mesquite West Study Area

A summary report on the SWF along the Lower Colorado River and tributaries (McLeod and Pellegrini 2013) described the Mesquite West study area as situated within the Virgin River floodplain in Mesquite, Nevada (Fig. 1), with suitable SWF habitat at this site consisting of dense mixed-native stands of coyote willow and tamarisk amidst cattail and bulrush marshes. While the SWF do not nest in cattail marsh, they frequently nest near cattail marshes (Unitt 1998) and build nests with the cattail tufts, grasses, and shredded bark. The cattail marsh,

moreover, sustains insect populations and would provide a prey base for the SWF. The insectivore forages within the habitat, above the canopy, above water, and along the patch edge, mostly by aerially gleaning an insect from trees, shrubs, and herbaceous vegetation (Sogge et al. 1997).

In this study, the cattail marsh serves as a single indicator for wetland identification and surrogate for soil moisture since no data on soil moisture or groundwater exist for the site because of its obligate wetland indicator status and easily recognizable signature. Study definition of SWF habitat in the Mesquite West entails all living vegetation features essential for breeding, nesting, foraging, rearing of young, and cover for climatic and predatory protection.

Surveys from 2003-2012 detected 6-30 resident adult SWF annually. Irrigation return flows and human manipulation of the channel influence the hydrological conditions of the site. In the years 2003-2008 and in 2010, standing water and muddy soils were noted throughout the breeding seasons; however, in 2009 and in 2011, a disruption of the normal irrigation return flows occurred, and water only inundated the eastern portion late in the breeding season in 2009 and bypassed the site altogether in 2011 due to channel dredging. By 2012, after construction of a berm to redirect return flows to the northeastern corner of the site, the site experienced intermittent inundation and reduction of area with standing water. In addition, the winter flood in 2005, a natural disruption, altered the channel course of the Virgin River and scoured the southeastern section of the site. These kinds of changes, either caused by stand maturation, natural events, or anthropogenic activities, are expected to occur in the future. With the dwindling populations of the SWF, it is crucial to understand changes over time and conserve its habitats.

FIGURE 1

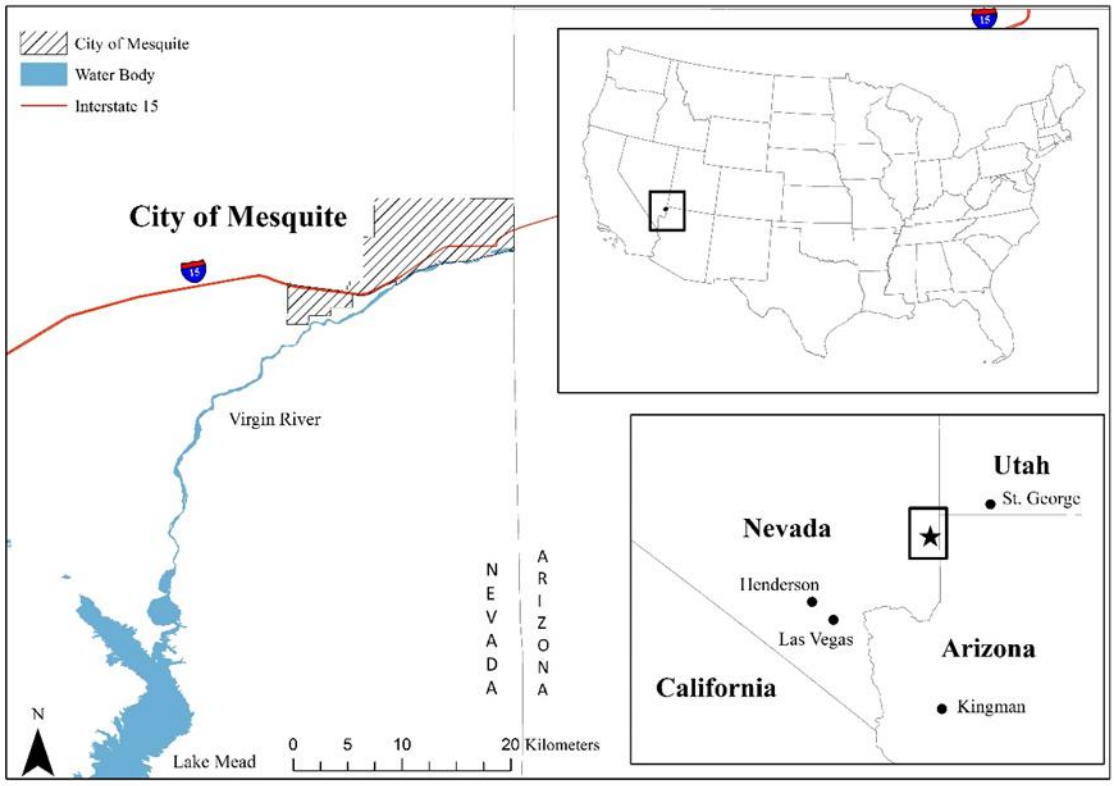

Figure 1: Geographic location of Mesquite, which is 128.8 km (80 miles) northeast of Las Vegas in Clark County, Nevada.

Field observations in 2012 (Fig. 2) and current Google Earth imagery indicate the area has changed: absence of surface and standing water in the habitat observed in 2004 and less dense vegetation with more open canopy.

FIGURE 2

Figure 2: Standing water observed in previous years had disappeared by the 2012 field visit to the Mesquite West site. Note the hexagonal desiccation cracks from the shrinkage pattern caused by the reduction of water content and drying of the soil materials. Photograph by Teri Jacobs (June 2012).

MATERIALS AND METHODS

The U.S. Bureau of Land Management (BLM) supplied orthorectified CIR aerial photography for the years 2004 and 2010 for Mesquite West (0.305-m and 1-m resolution, respectively). A commissioned flight in 2004 during August under bright sun (sun angle > 30°) and cloudless conditions captured images of the site in August by 3001 Inc., using a Z/I Digital Mapping Camera (DMC) with on-board Global Positioning System (GPS) and IMU technology. While missing

information on the bandwidth, the 2004 CIR aerial photography has three bands, green, red, and near-infrared (NIR), and a spatial resolution of 0.305 m at a scale of 1:600. For the 2010 CIR aerial photography, the National Agriculture Imagery Program (NAIP) flew a standard mission to acquire 1-meter spatial resolution (1:2,000) imagery in the growing season, specifically June for the site, during optimal atmospheric conditions with DMC systems. The nominal focal length of 120 mm of the digital frame camera projected an image on virtual charged-coupled devices (CCD) measuring 13,824 by 7,680 pixels with a pixel size of 12 μm by 12 μm. Four 3,072 by 2,048 pixel multispectral (MS) cameras with 30 mm lenses produced the red, green, blue, and NIR images (bandwidth information unknown).

Image acquisition, as noted previously, occurred during optimum daylight conditions on August and June. Ideally, the images should be taken on or near the same date, or at least the same season, to minimize differences in phenological cycles and sun angles. While not representative of the optimal anniversary dates, the image acquisition dates do match in season, summer, when the vegetation is mostly phenologically stable (Singh 1989; Coppin et al. 2004).

The change detection approach is an integration between processing in GIS and remote sensing. First step involves pre-processing in GIS, then vegetation delineation in ENVI, followed by the quantification and visualization of the changes within GIS. In preparation for the hybrid change detection, we performed image resampling and subsetting in ArcMap 10.1 (ArcGIS® software Environmental Systems Research Institute, Redlands, California) to rectify the differences of spatial resolution and extent between the images. Resampling the pixel sizes of the highest resolution image to the coarsest resolution entailed the use of the resampling tool in the Raster Processing toolset under Data Management. We opted for the nearest neighbor resampling technique after assigning 1-meter for the cell size. To amend the difference in spatial extent between the images, we created an image subset, a section of the larger image that enables the elimination of non-target objects, such as the urban features, to focus strictly on the area of

interest, the SWF habitat. As a distinction from other SWF habitat research,, this study does not restrict SWF habitat to nesting habitat and defines the SWF habitat at the Mesquite West survey site as all live green foliage (Allison et al. 2003) within the image subset. We accomplished this task with the proprietary 2004 SWF data provided by the U.S. Bureau of Reclamation (BOR) by ensuring all point locations of nests and presence detections of resident SWF were contained in the digitized polygon mask used to extract a subset of the image raster.

Change detection of SWF habitat relies on the ability to discern live green foliage. For this capacity, we chose NDVI. By differentiating between green and non-green surfaces using the ratio of red and near-infrared bands (NDVI = [*NIR – RED*]/ [*NIR+ RED*], where NIR refers to near-infrared band reflectance wavelengths of 750-1300 nm and RED refers to the red band reflectance wavelengths of 600-700 nm), NDVI is widely used to correlate strongly with plant condition, leaf area index, biomass, and vegetation cover. The application of NDVI to monitor temporal changes in vegetation has been well documented (Carlson and Ripley 1997; Lyon et al. 1998; Nagler et al. 2001; Kerr and Ostrovksy 2003; Lunetta et al. 2002; Lunetta et al. 2006; Manciono et al. 2014; Ghandi et al. 2015; and others). Nagler et al. (2001) assessed three common vegetation indices and found the NDVI best predicted percent of vegetation cover. In a comparison of seven different vegetation indices used for change detection, Lyon et al. (1998) concluded the use of NDVI produced more accurate difference image results than the other vegetation index groups because topography affected it less and the estimations were more consistent with visual interpretations and field work.

In ENVI 4.7 Spectral toolbox, the SPEAR Vegetation Delineation tool calculated the NDVI image for analysis. Two earlier studies incorporated the Vegetation Delineation tool to assess vegetation stress caused by beetle infestation (Filchev 2012) and to extract vegetation

from an urban scene (Rahman 2014). Other SPEAR tools described in the literature included two studies utilizing the Destriping Tool (Scheffler and Karrash 2013; Hamadache et al. 2014) and one that introduced the Change Detection Wizard as a new wetland monitoring approach (White and Lewis). Since the Change Detection Wizard required the same number of bands for the analysis, an assumption our imagery data violated, we could not employ this approach and opted to assess the presence of vegetation and level of vigor with the Vegetation Delineation Tool.

The image processing flow of the SPEAR Vegetation Delineation Tool consists of four steps, five if you choose the option to perform a spatial subset: (1) input image file and optional spatial subset; (2) atmospheric correction; (3) NDVI calculation; and (4) examine results.

For the atmospheric correction, the user may select none/already corrected, dark object subtraction, flat field correction, internal average relative reflectance, log residuals, or empirical line calibration. The best method depends on skill level, scene content, desired application, and available ancillary data (e.g., spectral library with signatures for materials contained in the scene). The most accurate method for both spectral shape and absolute reflectance values is the empirical line calibration; however, this method requires ground truth knowledge. For best balance between accuracy and simplicity, users would opt for dark object subtraction.

The NDVI calculation prompts the user to select the appropriate bands for the NIR, red, blue, and green inputs. As a result, this generates four default display ranges: (1) “No Veg”; (2) “Sparse Veg”; (3) “Moderate Veg”; and (4) “Dense Veg”. The numbers within the range indicate the bottom NDVI threshold. NDVI values range from -1 to 1, where higher values indicate greater chlorophyll density and healthier green vegetation (Rahman 2014; Gandhi et al. 2015). In general, dense vegetation have NDVI values nearing one (0.6 to 0.9), sparse or moderate

vegetation have moderate NDVI values (0.2 to 0.5), barren areas of rock, soil, or snow have NDVI values nearing zero (0.1 or less), and water bodies have negative NDVI values. The Density Slice in the SPEAR Vegetation Delineation tool enables the user to flicker, blend, or swipe between an overlay of the false color image and the base NDVI image and manipulate the thresholds for the vegetation classes. With this ability, analysts have the advantage of manipulating the default ranges to suit the purposes of a particular study. Live green foliage, whether dense, moderate, or sparse, mattered in this study, and we reclassified the image into one class with the bottom NDVI thresholds 0.25 (2004) and 0.32 (2010) to capture as much live green foliage as possible with the tool and exported the vegetation mask as a shapefile.

In ArcMAP 10.1, we imported the "Vegetation" shapefile and clipped with the mask. A field was added to the attribute table and the geometry for the area was calculated. By differencing the area of the 2010 vegetation mask from 2004 vegetation mask, we quantified the vegetation cover changes.

We created an overlay map, using color-blind friendly primary palettes red and yellow for the 2010 and 2004 scenes, respectively. A transparency of 30% applied to the 2004 image, thereby permitting the colors to blend and produce a secondary color orange for the overlapping areas to aid in the visual change detection analysis.

As a guide for replication, a detailed step-by-step graphic follows (Fig 3).

FIGURE 3

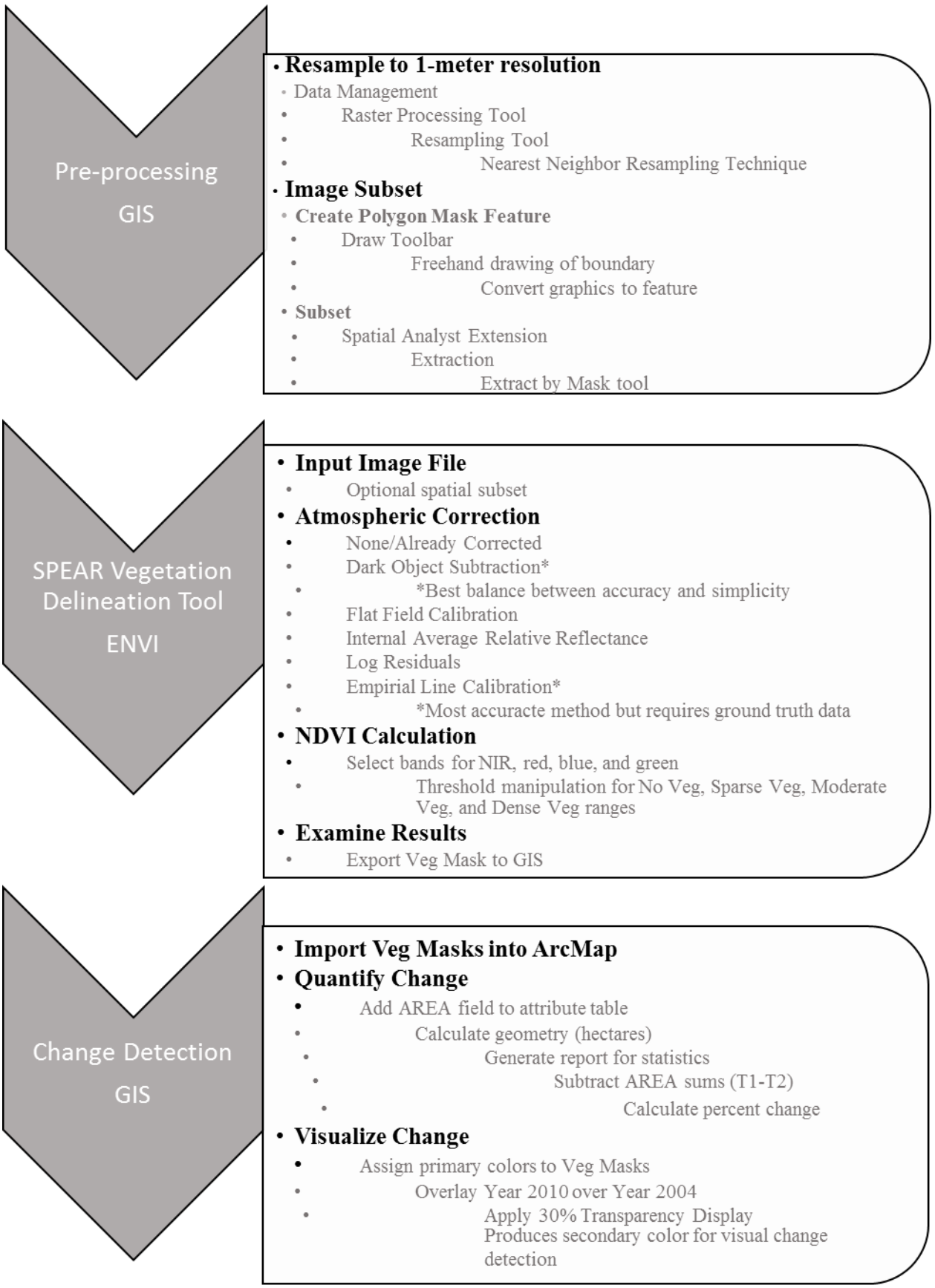
Pre-processing
GIS
• Resample to 1-meter resolution
• Data Management
• Raster Processing Tool
• Resampling Tool
• Nearest Neighbor Resampling Technique
• Image Subset
• Create Polygon Mask Feature
• Draw Toolbar
• Freehand drawing of boundary
• Convert graphics to feature
• Subset
• Spatial Analyst Extension
• Extraction
• Extract by Mask tool
SPEAR Vegetation Delineation Tool
ENVI
• Input Image File
• Optional spatial subset
• Atmospheric Correction
• None/Already Corrected
• Dark Object Subtraction*
• *Best balance between accuracy and simplicity
• Flat Field Calibration
• Internal Average Relative Reflectance
• Log Residuals
• Empirial Line Calibration*
• *Most accuracte method but requires ground truth data
• NDVI Calculation
• Select bands for NIR, red, blue, and green
• Threshold manipulation for No Veg, Sparse Veg, Moderate Veg, and Dense Veg ranges
• Examine Results
• Export Veg Mask to GIS
Change Detection
GIS
• Import Veg Masks into ArcMap
• Quantify Change
• Add AREA field to attribute table
• Calculate geometry (hectares)
• Generate report for statistics
• Subtract AREA sums (T1-T2)
• Calculate percent change
• Visualize Change
• Assign primary colors to Veg Masks
• Overlay Year 2010 over Year 2004
• Apply 30% Transparency Display
Produces secondary color for visual change detection

Figure 3: Flow chart showing the major analysis steps and the analysis tools and tool parameters nested beneath.

We performed maximum likelihood supervised classification and iso-cluster unsupervised classification using the Image Classification extension in ArcMap as a qualitative way to validate and compare method results (for classification details, see e.g., Ozesmi and Bauer 2002; Jensen, J.R. 2007; Xie, Sha, and Yu 2008; Perumal and Bhaskaran 2010). For each technique, we assigned pixels to four classes—riparian vegetation, unhealthy vegetation, sediment, and water—and extracted the riparian vegetation class from the classification image results for comparison.

RESULTS

The map (Fig.4) illustrates the visual difference of the change detection analysis. Areas in red represented vegetation present in 2010; yellow represented vegetation present in 2004; and orange represented vegetation present in both 2004 and 2010.

FIGURE 4

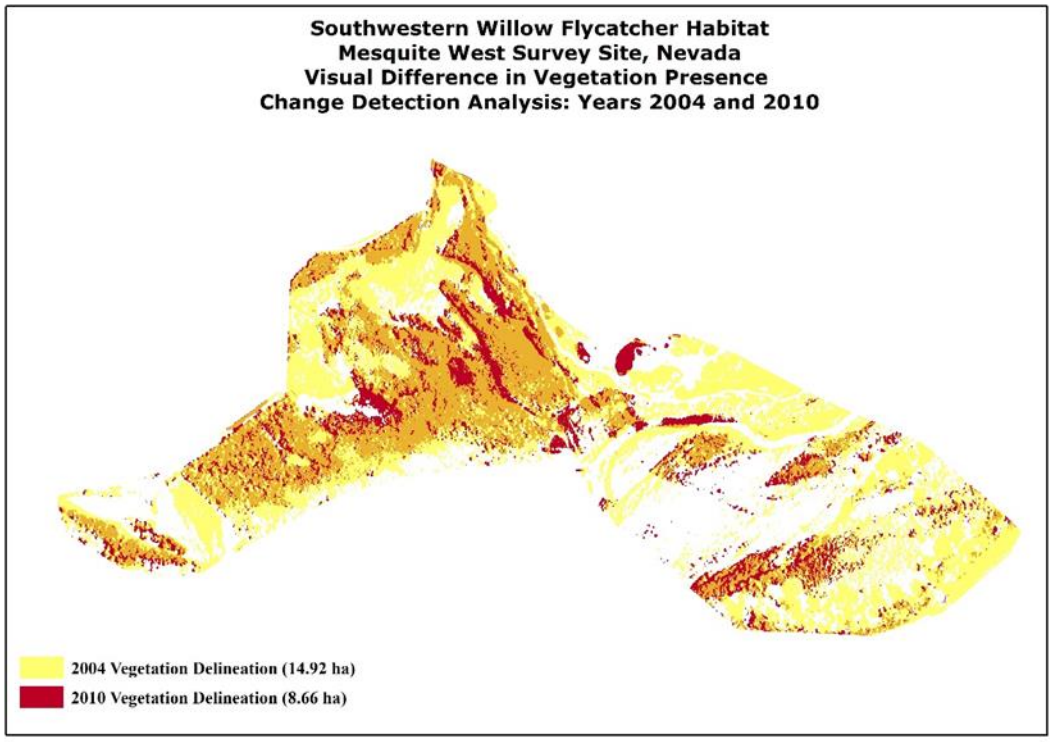

Figure 4: Visual difference in vegetation presence can be seen in the map to detect changes between 2004 and 2010. Orange denotes the overlapped areas, while yellow represents vegetation present only in 2004 and red represents vegetation present only in 2010. To protect the Mesquite West breeding site, all identifiable location features have been removed and only the vegetation polygons remain.

In 2004, the riparian vegetation, including cattail marsh and shrubs, measured 14.92 hectares, but by 2010 with a total area of 8.66 hectares, the vegetation cover diminished by 6.26 hectares. This constitutes a 42% decrease in vegetation cover and available habitat for nesting, foraging, and patrol perches for the SWF. The graph in Figure 5 shows the decline.

FIGURE 5

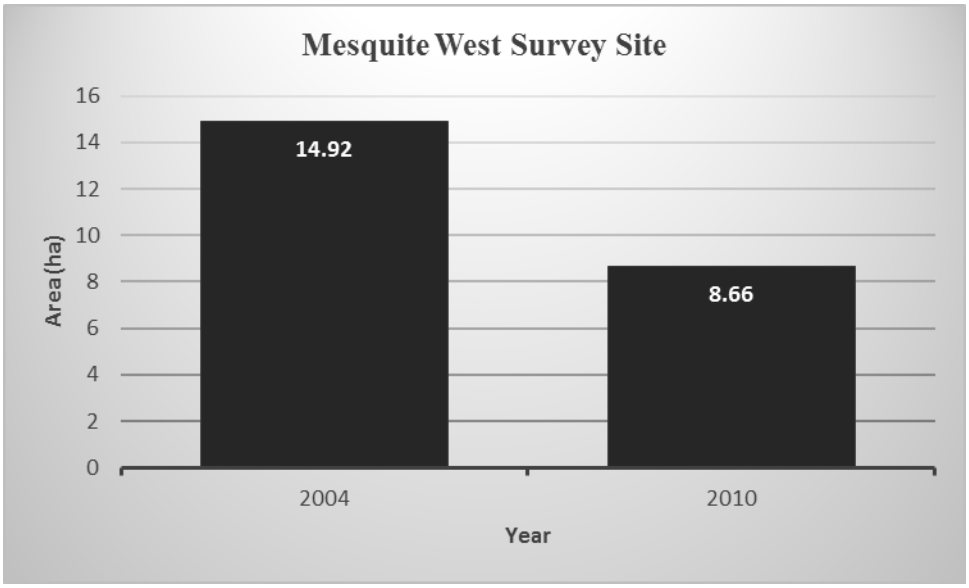

Figure 5: The column graph depicts the areas of vegetation cover in 2004 and 2010.

Visual interpretation of the change detection results map confirms the loss of the cattail marsh in the northwestern region in the study area by 2010; additionally the lack of marsh in the west and surface water in the golf course to the east further evince drier conditions in 2010. In the southeastern section, we can observe the scouring and deposition effects of the 2005 winter flood and the altered channel course of the Virgin River. Vegetation growth in 2010 appears mostly in areas where surface water occurred in 2004, in some of the scoured section, and the nearby the new channel course.

Visual analysis of the three techniques per year reveal marked differences in the riparian vegetation cover (Fig. 6) and how well the vegetation classification/delineation matched the color-infrared aerial photograph (Fig. 7). In Figure 8, the graphic illustrates a declining trend in vegetation cover for each technique, but the Unsupervised Classification resulted in a less obvious trend (11% decrease in vegetation cover). The SPEAR Vegetation Delineation tool and

the Supervised Classification had comparable results in vegetation cover area change—42% decrease versus a 37% decrease in vegetation cover, respectively.

FIGURE 6

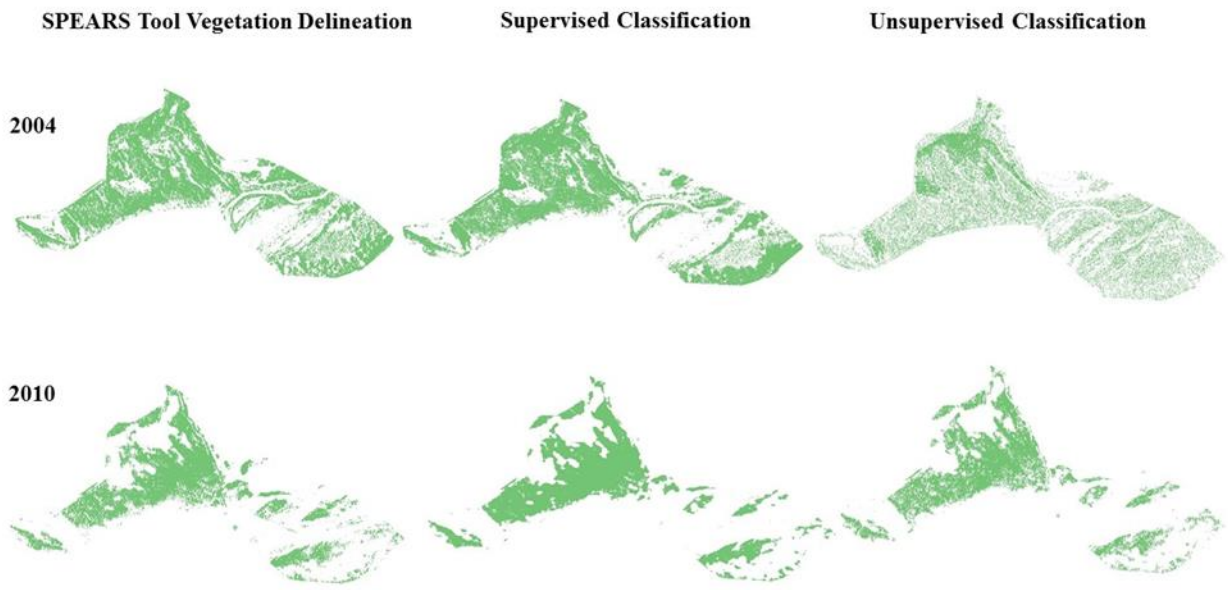

Figure 6: Vegetation cover results from the three techniques (columns) for each year (rows).

FIGURE 7

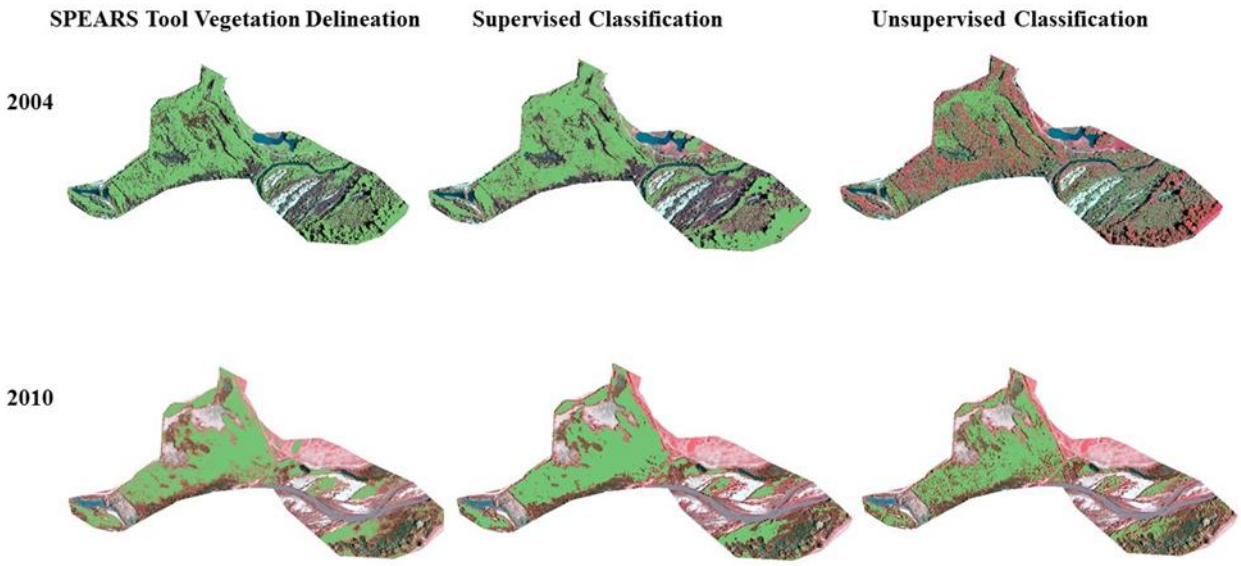

Figure 7: Overlay mapping results of the vegetation cover and false color images of the study site for each technique (columns) and each year (rows).

FIGURE 8

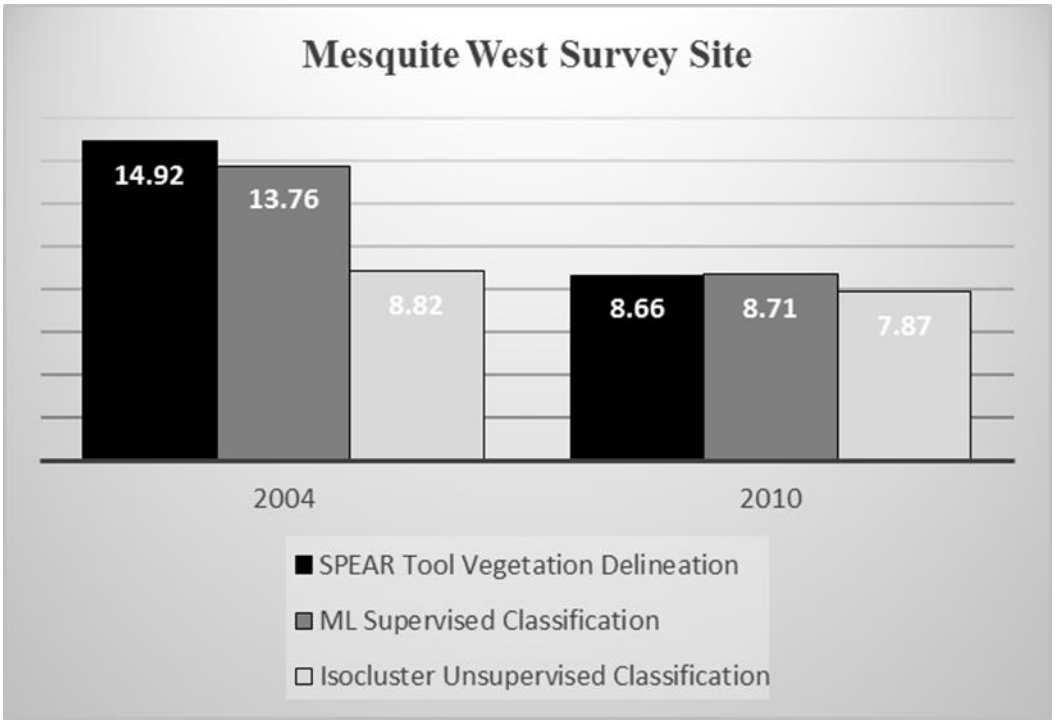

Figure 8: The column graph depicts the areas of vegetation cover in 2004 and 2010 by each technique: SPEAR Vegetation Delineation Tool, Maximum Likelihood Supervised Classification, and Isocluster Unsupervised Classification.

DISCUSSION AND CONCLUSION

Mesquite West Habitat Changes and Implications for Conservation

From an ecological perspective, change detection analysis can elucidate understanding of the spatially and temporally dynamic hydrological conditions and vegetation patterns of SWF riparian habitat. SWF riparian habitat fluctuates seasonally in response to precipitation, water management, and surface runoff (Paradzick and Hatten 2004) and is subject to periodic disturbance (Marshall and Stoleson 2000). Any perturbation of the underlying processes will lead to changes in the distribution, abundance, and composition of riparian habitats. Types of impacts for known habitat losses include fire, human activities such as agriculture or

construction, reservoir inundation, flood control efforts, and drought (Marshall 2000). These changes affect suitability for SWF occupancy and nesting (McLeod and Pellegrini 2013) and, consequently, SWF populations. Furthermore, in sites with more open canopy and less dense vegetation, brown-headed cowbird (*Molothrus ater*) parasitism of SWF nests increases and reduces SWF reproductive success (Uyehara et al. 2000). Population growth rates decrease with increased parasitism (Uyehara et al. 2000). At sites where habitat loss has already reduced SWF populations, parasitism may be the final driver toward local extinction.

It is imperative to document habitat losses. This study represents the first attempt to map SWF riparian habitat changes over time at the Mesquite West site. Since biologists only began surveys in 2003, we do not know the normal variation of vegetation response to irrigation flows, precipitation, groundwater, or flooding events.

The 42% decline in riparian vegetation at Mesquite West does indicate changes in the wetland hydrology. Observations of the cattail marsh in 2004 confirm the presence of hydric soils, and its disappearance by 2010 signifies the absence of the conditions that form hydric soils, typically inundation due to shallow standing water throughout the year or at least seasonally flooded and saturated. These findings align with survey results conducted at Mesquite West, whereby dry conditions and poor nesting success were recorded in 2009 (McLeod and Pellegrini 2013).

Disruptions in the site's hydrology appear to have affected SWF fecundity and productivity. When irrigation flows were diverted from the site in 2009, productivity and fecundity were very low but upturned in 2010 after irrigation flows returned to the site (McLeod and Pellegrini 2010). The mean fecundity and productivity again plummeted the following years when the site no longer experienced inundation, indicative of a need to manage irrigation flows

for the protection of the habitat and local survival of the critically endangered species. A steady or growing population defines the suitability of a site (Sogge et al. 1997). At Mesquite West, the vegetation changes may have caused the steady decline and may cause the potential extirpation of SWF populations, if this has not occurred already. The numbers of females have declined steadily since 2006, whereby each year had fewer and fewer females until only 1 was present in 2013 (McLeod and Pellegrini 2013). While biologists do not know the minimum viable population for SWF (Unitt 1998) or how long sites without visibly saturated soil or standing water will continue to support the riparian vegetation and/or remain occupied by SWF (Sogge et al. 2007), the implications should be concerning nonetheless.

Regional population dynamics depend on the small isolated sites, such as Mesquite West (Marshall 2000). Access to other neighboring populations along the Virgin River and Muddy River promote connectivity, gene flow, and metapopulation stability, and the distribution of suitable habitat should exist for SWF movement within its range. Unsuitability for breeding may not translate to unsuitable for foraging or use during migration. Therefore, even if SWF pairs are possibly absent from the site, the site should remain a conservation priority to maintain the riparian habitat in case of migration or future forced dispersals from local or larger sites (Marshall 2000).

We would recommend the use of this study's method to monitor the vegetation changes in all SWF territories. Changes in vegetation that equate to habitat loss will alert managers and prompt conservation action; changes in vegetation responses to rejuvenation will help managers evaluate the effectiveness of restoration projects. Monitoring will also provide information on the changes from natural cycling of SWF habitat. As habitats mature too much for SWF suitability, stochastic events clear older stands and create suitable environmental conditions for

revegetation from sediment deposition, seed dispersal, and groundwater recharge (Sogge and Marshall 2000; Allison et al. 2003; Ellis et al. 2009). Changes in riparian habitat may then be correlated with changes in SWF populations to apprise managers of the situation for appropriate responses. Ellis et al. (2009) surmised Roosevelt Lake management problems stemmed from the lack of information on the long-term effects of reservoir inundation on recovery ability of the SWF and the impacts of dams on wildlife demographics. With the steps outlined in the study, we may broaden ecological knowledge and fill gaps essential for conservation planning and management.

Potentials and Limitations of the Rapid Change Detection Using CIR Aerial Photography

The rapid global changes in landcover necessitate the development of fast, simple, and effective change detection techniques that meet the needs of NGOs, ecologists, biologists, conservation managers, policy makers, stakeholders, and other groups interested in modeling changes in biodiversity (Sader et al. 2001). This study delivers an innovative technique that produces results more quickly than any other known method. For the analysis of high-resolution digital aerial imagery, we accomplished this by employing the SPEAR Vegetation Delineation tool, which does not rely solely on subjective, expensive, and time-consuming visual interpretation (e.g., Heiskanen et al. 2008; Ihse 2008) or pixel-based or object-oriented classification schemes (e.g., Paradzick and Hatten 2004; Paxton et al. 2007, Ihse 2007; Gweon and Zhang 2008; Everitt and Yang 2010). Our change detection technique lacks the lengthy preprocessing and processing steps inherent in pixel-based and object-based interpretations. Furthermore, our technique produces better results than pixel-based classifiers, which can result in a salt-and-pepper effect with the use of high resolution images, and is more accessible than

object-based classifiers because of the difficulty associated with object creation and the less availability and affordability of the software (Morgan et al. 2010).

Application of the SPEAR Vegetation Delineation tool appears to have outperformed the traditional classification methods. Compared to the tool's vegetation delineation results, the thematic maps generated from the supervised and unsupervised classification schemes underestimated the amount of living green vegetation, one of the main criteria for SWF habitat suitability (Allison et al. 2003). The overlays of the vegetation class and false-color imagery of the site reveal more missed healthy vegetation—evident in the observation of red and bright pinks in false color—in the traditional classifications, especially the unsupervised classification in 2004 CIR aerial photography. The SPEAR Vegetation Delineation tool captured more healthy vegetation than the traditional classification schemes, which substantiates our supposition that the tool outperformed the traditional classification. Of note, the measurements of area differed more between the techniques with the 2004 CIR aerial photography than the 2010 CIR aerial photography. The texture variation between the scenes accounts for this difference, attesting to the difficulty of performing pixel-based classification on high resolution aerial photography of riparian habitats with high spatial and spectral heterogeneity. The 2004 unsupervised classification, indeed, resulted in a colorful salt-and-pepper appearance, a common problem with pixel-based classification (Whiteside and Ahmed 2005). In the 2004 scene, the forest canopy is more extensive and non-uniform with taller, more mature tree stands intermixed with shrubs and cattail marsh, imparting a rough texture. Whereas, in the 2010 scene, the areas with revegetation after the scouring 2005 flood appear more uniform and large patches of sandy substrate are visible throughout, lending it an overall smooth texture. This led to misclassification. Accuracy in classifying vegetation for change detection purposes is important because any errors in the

classified image amplify errors in the post-classification change detection process and lead to false changes in any direction or location (Singh 1989).

We consider our approach an appropriate method and recommend it over the traditional pixel-based classification methods for delineating riparian vegetation in a semi-arid environment and performing change detection analyses using high-resolution imagery for the main three following reasons:

1. ***Easy to implement***—even a novice remote sensing analyst could access and implement the tool. While conservation scientists recognize the power and benefit of remote sensing and GIS for ecological spatial analysis and modeling (Kushwaha and Roy 2002; Kerr and Ostrovsky 2003; Turner et al. 2003; Alpin 2005), not all ecologists or conservation biologists have advanced training in the geospatial technologies. Having uncomplicated, efficient methods is paramount in conservation research, where time is of the essence. The tool of this study achieved its objective to introduce such a method.
2. ***Semi-automated process***—the analyst could use the default NDVI thresholds or manually revise the thresholds to suit specific needs, such as in our case. Often the variability in aerial photography acquisition impede the use of automatic digital methods (Heiskanen et al. 2008).
3. ***GIS capable***—ability to export to GIS as a layer to quantify area and map change detection results. While aerial photography is the primary data for vegetation delineation, the use rarely involves quantitative measurements (Fensham and Fairfax 2002).

Aerial photographs provide the longest-available, temporally continuous, and spatially complete record of landscape change; however, digital aerial photography has a shorter time range (since the 1990's) (Morgan et al. 2010). For this reason, maps derived from aerial photography have routinely been used for ecosystem management and decision-making (Cohen et al. 1996). The use of aerial photography to study and map wetlands has a long history with a multitude of applications: identifying wetlands, classifying plant composition, estimating productivity and/or abundance, assessing quality, delineating wetland ecotones, disturbance and invasive species mapping, and change detection. Given the high quality image and fine detail, aerial photography provides geometrically and visually accurate representations of the visual scene. Aerial photographs contain a spectrum of useful information. Color/tone, shape, size, shadow, pattern, texture, and association (site and content) aid in the photointerpretation of the key characteristics of ecological features. In this study, the ability to discern live vegetation from dead/dying vegetation by color (e.g., in false color images, green or light pink indicates low plant density, whereby the tones of the soil showed) (Statewide Mapping Advisory Committee, 2011) facilitated the visual interpretation of the NDVI threshold and should translate to higher accuracy despite the limitation of not having ground truth data to validate results.

A major limitation of the research hinges on the inability to conduct an accuracy assessment due to the lack of ground truth data. Ideally, we would want to involve field work and collection of GPS points for ground truth data at the start and end dates to assess accuracy, but, in the absence of in situ ground control points for accuracy assessment, generally the change detection output image is compared to a higher quality information source (Stehman 2009). The study's CIR aerial photography at 0.3 m resolution, however, has no readily available, cost-effective comparison. We cannot ascertain validity of results but assume sufficient accuracy.

Experts regard aerial photointerpretation of wetland vegetation analysis and mapping as reliable, repeatable, consistent, accurate, successful, and superior to satellite imagery (Tiner 1996; Fensham and Fairfax 2002; Valta-Hulkkonen et al. 2005; Ihse 2007; Yang 2007; Morgan et al. 2010 and others). The use of aerial photography to assess satellite data implies accuracy (Fensham et al. 2002).

Still, errors beyond our control may have occurred during flight, camera calibrations, or orthorectification. We acknowledge the topographic shadows and other imagery anomalies in the aerial photographs may have contributed to analyst errors in omission and commission during the vegetation delineation, whereby the NDVI threshold failed to capture all the living biomass precisely. Despite NDVI's ability to reduce multiplicative noise, such as illumination differences, shadow, atmospheric attenuation, and some topographic variations (Huete et al. 2002; Chen et al. 2005), some deficiencies associated with NDVI may need corrected (Jones and Vaughn 2010). The surface soil and water reflectances may have affected the NDVI values of the canopy and the apparent "greenness" of the vegetation in the study's images. An experimentation with another index, such as the Normalized Water Index (NDWI) used in a recent wetland mapping and change detection study (Kavyashree and Ramesh 2016) or other modification of the index would redefine the vegetation delineation and enhance change detection performance. In our study, NDVI's inability to detect the relative contributions between grass from the golf course and tree canopy (Parrini, Macindoe, and Erasmus 2013) confounded the delineation of the riparian vegetation.

The higher cost of aerial photography may also hinder its use for some wetland mapping and analysis, necessitating the alternative use of cost-effective satellite imagery. In two previous studies on change detection of regional swaths of SWF habitat, Paradzick and Hatten (2004) and

Paxton et al. (2007) developed methods using Landsat TM+ imagery, NDVI, and probability classes. Landsat satellite imagery does have the advantage of cost-free open availability, large archive, continuous record, and future repeatability; however, we do not know if the application of this method to moderate-resolution data would be as successful, which requires additional study to determine its feasibility. Riparian vegetation in the Southwest appear as very thin green ribbons along the stream system in Landsat imagery, and the coarse detail and scale for a may be inappropriate for effective change detection analysis. If so, then the necessity of aerial photography or high-resolution satellite data for monitoring changes in small riparian ecosystems warrants the costs.

Further application of method may include employing multi-sensor data. Various sensor types differ in spectral, spatial, radioactive, and temporal characteristics and deliver different details about the environment. For this reason, a certain type of sensor produces imagery better suited for some purposes but not others (e.g., weather data generated every half hour from NOAA's GOES satellite). The combined use of multi-sensor data with advanced methods, though, can enrich understanding of ecological features, processes, and patterns and improve classification results (Pohl 2016). However, most change detection techniques were designed to process imagery acquired from the same sensor or sensor type (Pillai and Vatsavai 2013). The increasing number of air- and space-borne sensors and the difficulty in obtaining same sensor data for a particular location over time have triggered a need for techniques that handle multi-sensor data (Alberga 2009; Lu et al. 2010; Forkuo and Frimpong 2012; Pillai and Vatsavai 2013). The computational complexity and ad-hoc target data fusion methods (Volpi et al. 2013) may hinder the use by many applied ecologists and conservation biologists who lack training. To which, this study's proposed method may alleviate and be more broadly accessible.

Another direction for future research would focus on testing this method on temperate landscapes. In a semi-arid environment, riparian vegetation stands out a vibrant green amongst the sandy and rocky barrens; this distinct contrast simplifies the delineation. The boundary between wetland and upland in humid regions, however, is not clear-cut, thereby making the delineation more difficult. Still, even in more humid regions, wetland vegetation has higher NDVI values than surrounding upland vegetation. Distinguishing wetland vegetation with the NDVI thresholds, whether by a priori knowledge or experimentation, in the SPEAR tools should be effective in humid regions as well.

In conclusion, the novel methodology presented here proved suitable for change detection of the designated critical riparian habitat in a semi-arid environment for the federally listed endangered SWF. The results of this study implicate a need for management to insure the continuance of Mesquite West as a breeding territory for the endangered SWF. The SWF report recommended restoration efforts along the Virgin River to provide alternative nesting territories, maintenance of existing breeding sites, and continued dialogues with landowners, the Mesquite Irrigation District, and the Bunkerville Irrigation District (McLeod and Pellegrini 2013). Recovery might be possible if these community entities maintain irrigation flows toward the breeding site and concentrate on watershed management of upland, headwaters, main river stem and tributaries. Otherwise, the fate of the SWF population at Mesquite West may end the same as the eight territories along the middle Rio Grande in New Mexico: abandonment (Marshall and Stoleson 2000).

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CHAPTER FOUR

Multi-Sensor Change Detection: An Introduction to an Integrated Remote Sensing and Geographic Information Systems Approach Using High-Resolution Data

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ABSTRACT

Habitat loss, degradation, modification, and fragmentation threaten biodiversity worldwide. In recognition of the importance of biodiversity and growing crisis, conservation efforts prioritize the monitoring of biodiversity. Remote sensing provides the only effective means in terms of cost and time in the acquisition of accurate ecological data and realistic monitoring of landscape changes, and its use has demonstrated high value for conservation purposes. However, change detection techniques were designed to process imagery data acquired from the same sensor or sensor type. With the increasing number of air- and space-borne sensors and the difficulty in obtaining same-sensor data, the need to develop change detection techniques to handle multi-sensor data has arisen. The recently developed methods are computationally complex, which limits their use, and few studies address this problem in a readily available way for non-experts in remote sensing. In response, this study expands upon a previous study to investigate the effectiveness of an approach integrating the Spectral Processing Exploitation and Analysis Resource Vegetation Delineation and Stress Detection Tool in ENVI software, which determines the level of vegetation vigor with the Normalized Difference Vegetation Index, and Geographic Information Systems to perform multi-sensor data change detection. We compared the approach introduced in the paper to the traditional supervised and unsupervised classification of vegetation at the site to illustrate the differences in wetland delineation, a challenging endeavor due to the inherent arbitrariness and indistinctness of boundaries. The results confirm a declining trend in the riparian habitat and imply the vegetation delineation tool more accurately identifies vegetation. The study delivers a change detection technique that can successfully handle the problem of data acquired from different high resolution sensors and provides a simple, time-effective, accurate, and reliable means to monitor environmental changes. For the non-expert in remote sensing, it is an advantageous tool monitoring of landscapes and ecosystems for habitat loss, degradation, modification, or fragmentation.

Keywords: Change detection, conservation, geographic information system, multi-sensor, NDVI, remote sensing, riparian ecosystems, Vegetation Delineation and Stress Detection Tool

Introduction

Habitat loss, degradation, fragmentation, modification, and other damaging influences threaten biodiversity worldwide. In recognition of the importance of biodiversity and its growing crisis (Grehan 1993; Brooks et al. 2006), conservation efforts, therefore, prioritize the monitoring of biodiversity on all levels—from genes to species to ecosystems—and their changes (Lawley et al. 2016; Willis 2015; Pettorelli et. 2014; Corbane et al. 2014). Scientists have traditionally collected the relevant biophysical data for this purpose from ground-based methods. However, remote sensing provides a realistic tool for cost- and time-effective acquisition of ecological data and monitoring landscape changes, and its use and value for conservation has been reviewed comprehensively (see the work of Pettorelli, Safi, and Turner 2014; Alpin 2005; Kerr and Ostrovsky 2003; Turner et al. 2003).

Change detection analysis involves a remote sensing process to identify differences between observed phenomena at different times. Sensors on board aircraft or satellites capture the electromagnetic radiance (EMR) emitted from the surface and measure the brightness of the target object. As a basic theoretical concept, change detection analysis assumes the EMR values reflect biophysical properties; therefore, a change in the land cover translates to a change in EMR values (Singh 1989). The applications vary widely: deforestation; urban sprawl; industrial development; coastal zone changes (e.g., erosion); climate change effects (e.g., sea ice, thaw lakes, glacial mass); farmland loss; crop monitoring; desertification; flooding; soil erosion; plant community changes; wetland changes; forest fires; forest mortality, defoliation, damage estimations; algal blooms; invasive species spread; etc. (see e.g., Lu et al. 2004). Assessments of land cover change attempt to explain where change occurs, the type of change occurring, the rate of change, spatial distribution of the change, and the probable cause/s for change. Moreover,

these assessments have increased in importance due to the magnitude and severity of global problems arising from land cover/use changes: desertification, eutrophication, acidification, climate change, sea-level rise, and biodiversity loss.

Remote sensing scientists have devoted enormous efforts into developing various change detection techniques, and many reviews of the methodologies exist in the literature (see e.g., Singh 1989; Coppin and Bauer 1996; Coppin et al. 2004; Lu et al. 2004; Hussain et al. 2013). However, most change detection techniques were designed to process imagery data acquired from the same sensor or sensor type (Pillai and Vatsavai 2013). Data ideally should come from the same sensor and have the same radiometric and spatial resolution with anniversary or near-anniversary dates to eliminate sun angle, season, and phenology differences (Lu et al. 2010).But, the difficulty in attaining same sensor data and the increasing number of air- and space-borne sensors has driven the need for the development of change detection techniques to handle multi-sensor data (Alberga 2009; Lu et al. 2010; Forkuo and Frimpong 2012; Pillai and Vatsavai 2013).

Most literature on change detection describe procedures on data acquired by the same sensor or type of sensor (Alberga 2009), and few describe multi-sensor change detection techniques (Millward et al. 2006; Volpi et al. 2013). There does not appear to be a comprehensive summary review on multi-date, multi-sensor change detection, despite multi-sensor change detection becoming more common due to limited availability of data from older or defunct sensors (Wulder, Butson, and White 2008). Despite criticisms of the errors inherent in post-classification for multi-sensor change detection, the post-classification method is often adopted (Ruelland et al. 2011). Post-classification for monitoring land cover changes, however, requires lengthy image pre-processing and image processing. Many of the other studies applied

transformations, generally image fusion (e.g., Pohl and Genderen 1998; Willbauck 2000; Gungor and Akar 2010) or NDVI (e.g., Millward, Piwowar, and Howarth 2006; Forkuo and Frimpong 2012; Mandanici and Bitelli 2015), prior to the pixel-based or object-oriented classification. Specific to wetland change detection, the spatial and temporal variability produce fine-scale mixtures of classes that may impede both classification and change detection, indicative of the development of approaches different than traditional landcover change detection (Muro et al. 2016).

For long-term change detection studies, researchers need methods to remedy the spatial resolution and thematic differences between multi-source imagery. This usually involves a combination of methods. More recently, researchers have introduced new multi-sensor change detection techniques, such as Modified Iterated Hough Transform (MIHT) used as a matching strategy for registration of images obtained from different sensors (Habib et al. 2005), similarity measures (Alberga 2009; Pillai and Vatsavai 2013), nonlinear kernel canonical correlation analysis (Volpi et al. 2013), and supervised classification of the iteratively reweighted multivariate alteration detection (IR-MAD) output (Pathak 2014).

This paper presents a simple solution for the multi-sensor change detection challenge, one that avoids lengthy processing and computational complexity inherent in the previously mentioned approaches. Previously, a survey of the riparian breeding habitats critical to the endangered southwestern willow flycatcher (Empidonax traillii extimus, hereafter referred to as SWF), comprised a case study on the small, isolated site in Mesquite, Nevada was conducted (Jacobs and Tong 2017, in press). An efficient approach was introduced to analyze change detection using color-infrared (CIR) aerial photography that significantly reduces processing time and enables researchers to not only produce maps identifying areas of vegetation change,

but to also quantify those changes, with specificity rarely attained using aerial photography (Fensham and Fairfax 2002). The technique is easy to implement in a semi-automated process and to export to Geographic Information Systems (GIS) for quantification and overlay analysis with sufficient accuracy. However, the question remains whether the use of the Spectral Processing Exploitation and Analysis Resource (SPEAR) Vegetation Delineation and Stress Detection Tool in ENVI software (Exelis Visual Information Solutions, Boulder, Colorado), which determines the level of vegetation vigor with the Normalized Difference Vegetation Index (NDVI), in combination with GIS, will perform as well using imagery data acquired from different sensors.

For the purposes of this experiment, the case study in Mesquite, Nevada, was revisited using not only CIR aerial photography but also Quickbird satellite imagery. This approach was evaluated (1) whether the rapid hybrid change detection introduced for CIR aerial photography worked as a feasible change detection technique for high resolution multi-sensor data and, if so, (2) whether the results indicated a declining trend in SWF habitat. As a consequence, this study will inform other remote sensing scientists and non-governmental organizations (NGOs) of the most time-effective, time-efficient, and reliable means to monitor changes in the environment for management and conservation objectives. This paper aims to serve as a bridge between remote sensing and ecological research.

Methods

Study Area

The city of Mesquite, Nevada, lies in the floodplain of the Virgin River about 128.8 kilometers (80 miles) northeast of Las Vegas, adjacent to the Arizona state line (Fig.1). Golf

courses, housing developments, and the Virgin River border the Mesquite West study site. In general four habitat types exist for the SWF: (1) monotypic high elevation willow (*Salix exigua* and *S. geyeriana*); (2) monotypic exotic, characterized by either salt cedar (*Tamarix* spp.) or Russian olive (*Elaenagnus angustifolia*); (3) native broadleaf dominated, often composed of single species Gooding's (*Salix goodingii*) or other willow, or mixed broadleaf trees and shrubs including cottonwood (*Populus* spp.), boxelder (*Acer negundo*), alder (*Acnus* spp.), and Ash (*Fraxinus* spp.); and (4) mixed native/exotic broadleaf trees and shrubs (Sogge et al. 1997). Mesquite West is primarily composed of the latter (Fig.2), with dense mixed-native stands of coyote willow and tamarisk situated amidst cattail and bulrush marshes, containing approximately 10.6 hectares (ha) and 6 to 30 resident adults annually as detected from 2003-2012 surveys (McLeod and Pellegrini 2013). With relatively moderate to very high canopy closure (50 to >90%), a mosaic of dense riparian vegetation patches interspersed with openings, acceptable tree height (5-6 m) for nests, and proximity to slow-moving or standing water, the Mesquite West site fits the criteria for suitable breeding habitat (USFWS 2002).

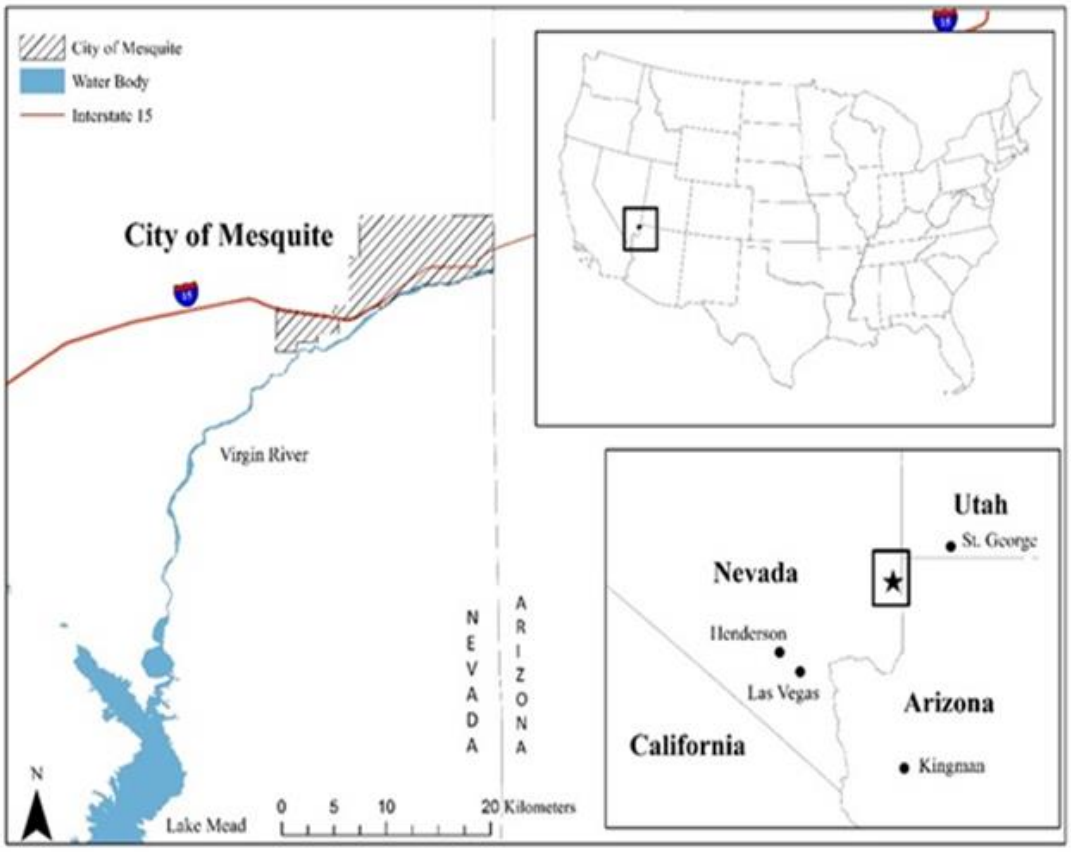

Figure 1: Geographic location of Mesquite, which is 128.8 km (80 miles) northeast of Las Vegas in Clark County, Nevada.

Figure 2: Image of a SWF nest amidst coyote willow at the Mesquite West site. Photograph was taken by Teri Jacobs (June 2012).

By definition, riparian vegetation requires substantial water. Hydrologic processes contribute to the formation, persistence, size, and function of wetlands, and, at Mesquite West, the irrigation flows and human manipulation of the channel influence the hydrological conditions. Observations of standing water and muddy soils occurred in the years 2003-2008 and in 2010, indicating suitable habitat characteristics for the SWF nesting season. Notable disruptions in the hydrologic processes included the flooding event in the winter of 2005 that scoured sections of the site and redirected the river channel, lack of irrigation flows in 2009, and the dredging of the channel in 2011 that resulted in all return flows bypassing the site. In 2012, the site experienced intermittent inundation, whereby standing water only reached a portion of the areas inundated in previous years, creating unfavorably dry habitat conditions as observed

during the field visit (Fig. 3). The insect prey base of the SWF depends on the presence of lentic water; consequently, lentic water is very important to the survival of the SWF.

Figure 3: Standing water observed in previous years had disappeared by the 2012 when the Mesquite West site was visited. Note the hexagonal desiccation cracks from the shrinkage pattern caused by the reduction of water content and drying of the soil materials. Photograph was taken by Teri Jacobs (June 2012).

While breeding habitats range from 0.6 ha to 100 ha in six states in the southwest (Finch 1999), nearly half of the total population occupies the smallest territories, such as Mesquite West, the distribution of which creates a conservation conundrum (Marshall and Stoleson 2000). The scarcity of suitable habitat is often the primary reason for the status of most rare and endangered species. These small SWF populations have low genetic variability and face the greatest risk of extirpation, and their persistence may depend on the connectivity among the patches and proximity to larger patches (Rocklage and Edelmann 2002). Protection of the

populations *and* ecological functions that sustain them requires landscape-scale approaches and management at the scale of the drainage basin, even in the existing small, isolated habitats.

Materials

The U.S. Bureau of Land Management (BLM) supplied the orthorectified 2004 and 2010 CIR aerial photographs (0.305-m and 1-m resolution, respectively) and the 2006 Quickbird satellite imagery (0.61-m resolution), and the U.S. Bureau of Reclamation (BOR) provided the proprietary SWF nest data from 2004 to 2010.

To minimize the differences in sun angle and phenological cycles, images should be taken on or near anniversary dates, or at least during the same season. The acquisitions of the three images during different months, while not representative of the ideal anniversary date, do match in season, during summer when vegetation is mostly phenologically stable (Singh 1989; Coppin et al. 2004) and coincide with the SWF breeding season.

Change Detection Analysis

As a first step in this hybrid change detection method, the differences in spatial resolution were rectified between the three images (2004 CIR aerial photography, 2006 Quickbird satellite imagery, and 2010 CIR aerial photography) by resampling the pixel sizes. The rule of thumb is to resample the higher resolution images to the coarsest resolution, which in this case is the 1-meter resolution of the 2010 CIR aerial photography. The second step entailed amending the differences of the spatial extents between the images with an image subset. An image subset is a section of the larger image that eliminates most non-target objects, such as the urban features, to focus strictly on the area of focus. In ArcMap 10.1 (ESRI 2017. ArcGIS Desktop: Release 10.2. Redlands, CA: Environmental Systems Research Institute), the SWF data on the locations of

nests and presence detections guided the determination of the focus area. A polygon mask feature was created by free-form drawing of the boundary around the SWF point data in the editing function and used for the raster subset extraction. Since precise territory perimeter data do not exist for this site, not all vegetation cover within the subset will equate to suitable SWF breeding/nesting habitat. The vegetation patches are considered "SWF habitat", regardless, for simplicity and ease of analysis.

The simple approach to the change detection process utilized the SPEAR Vegetation Delineation workflow in ENVI. Besides the whitepaper produced by ITT Visual Information Solutions, only two published articles have applied the vegetation delineation tool; these described the process to assess vegetation stress caused by beetle infestation (Filchev 2012) and to extract vegetation from an urban scene (Rahman 2014). In general, the SPEAR workflow toolset has been inadequately described in the literature (Hamadache et al. 2014). The only other examples consist of two studies incorporating the Destriping Tool (Scheffler and Karrash 2013; Hamadache et al. 2014) and one study introducing the Change Detection Wizard as a new wetland monitoring approach (White and Lewis 2011). While the SPEAR toolset includes a Change Detection Wizard with four types of algorithms (transform, subtractive, two color multi-view, and spectral angle), the tool recommends the use of images with the same viewing angle and requires the same number of bands for the analysis, the latter of which the available data violate. Therefore, the SPEAR Vegetation Delineation tool was employed to assess the presence of vegetation and the level of vigor within the three image subsets.

The SPEAR Vegetation Delineation Tool accomplishes this task by calculating the NDVI, which differentiates between green and non-green surfaces using the equation:

$$\text{NDVI} = \frac{\text{NIR} - \text{RED}}{\text{NIR} + \text{RED}}$$

where the NIR band has reflectance wavelengths of 750-1300 nm and the red band has reflectance wavelengths of 600-700 nm. The chlorophyll in leaves strongly absorbs visible light (400-700 nm) for photosynthesis, while the water-filled cells strongly reflect NIR light. Therefore, healthy plants with higher chlorophyll density have higher NDVI values; whereas, unhealthy or sparse vegetation reflect more visible light and less NIR light, which results in lower NDVI values. NDVI values range from minus one (-1) to plus one (+1), with dense healthy vegetation nearing one (0.6 to 0.9), sparse or moderate vegetation with moderate values (0.2 to 0.5), barren areas of soil, rock, or snow nearing 0 (0.1 or less), and water bodies in the negatives.

The SPEAR Vegetation Delineation tool consists of four steps in the image processing flow: (1) input image file and optional spatial subset; (2) atmospheric correction; (3) NDVI calculation; and (4) examine results. For the NDVI calculation, the Density Slice tool and visual interpretation of the false color base image enables the manipulation of the NDVI thresholds to classify the image by ranges. This ability is beneficial because the default ranges may not suit the purposes of a particular study, such as this one which needed only one class of vegetation, consisting of all the three subclasses: sparse, moderate, and dense, for the Veg Mask, the overlay parameter focused on vegetated pixels. Higher threshold values mask more pixels, and lower threshold values mask fewer pixels.

Once exported into ArcMap, the Veg Masks for each year were displayed using color-blind friendly primary palettes (yellow for 2004, blue for 2006, and red for 2010) and mapped with a 30% transparency setting on the top layer to permit the colors of the Veg Masks to blend

and produce secondary colors where no change occurred in vegetation between the years compared. Fields were added to the attribute tables to calculate area geometry. Quantification of the changes was accomplished by simply subtracting the total areas of vegetation cover over time—2006 from 2004, 2010 from 2006.

Accuracy Assessments

As a qualitative way to validate and compare the method results, a maximum likelihood supervised classification and an iso-cluster unsupervised classification were performed using the Image Classification extension in ArcMap. Five classes—riparian vegetation, unhealthy vegetation, grass, water, and sediment—were generated for each technique, with training classes assigned by the user in the supervised classification and with an iterative self-organizing procedure of class grouping by clusters and nearest means in the unsupervised classification. To compare to the vegetation delineation results, the classes were reclassified into binary groups of "live green vegetation" and "non-live green vegetation". Other papers describe these remote sensing image classifications in better detail (e.g., Ozesmi and Bauer 2002; Jensen, J.R. 2007; Xie, Sha, and Yu 2008; Perumal and Bhaskaran 2010). The riparian vegetation class was extracted from the classification images and used for visual comparison.

Quantification of image classification accuracy in remote sensing usually involves calculating a confusion matrix; however, without field-verified ground control points or pixels within the resultant vegetation delineation, an automated confusion matrix could not be performed. This study, therefore, explored the following alternative steps to calculate the confusion matrix:

- Generate random sampling points in ArcMap

- Classify random sampling points as either “live green vegetation” or “not live green vegetation” for each year, using the reference image data and each classification result
- Compare classifiers from the reference image data and classifications to identify the number of matches, false positives, and false negatives
- Manually calculate the confusion matrix to obtain the overall accuracy, the average producer’s accuracy, the average user’s accuracy, and Cohen’s Kappa coefficient that provides a more robust measure of agreement between the classified items than a simple percent agreement calculation.

Results

Visualized changes in the maps highlight the differences and similarities (Fig. 4 and 5). Figure 4 depicts vegetation present only in 2004 as yellow, vegetation present in 2006 as blue, and the areas of overlap, where vegetation occurred in both 2004 and 2006, as a pea-green color. The 2006 Veg Mask retains the blue color for the visual differencing with the 2010 Veg Mask in Figure 5, and the purple-indigo color represents the vegetation patches that occurred both in 2006 and 2010.

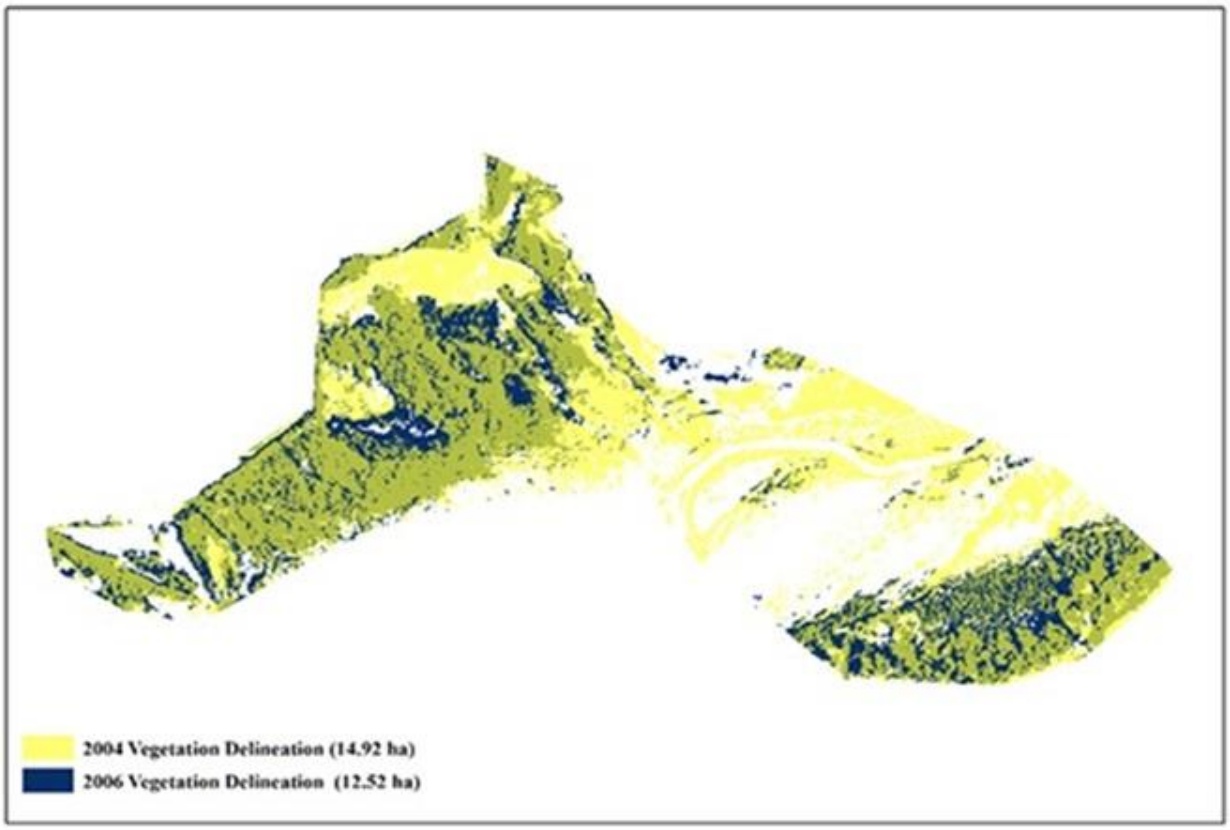

Figure 4: Southwestern willow flycatcher habitat at the Mesquite West survey site in Nevada. The difference in vegetation presence can be seen in the map to detect changes between 2004 CIR aerial photography and 2006 Quickbird satellite imagery. Green denotes the overlapped areas, while yellow represents vegetation present only in 2004 and blue represents vegetation present only in 2006. To protect the Mesquite West breeding site, all identifiable location features have been removed and only the vegetation polygons remain.

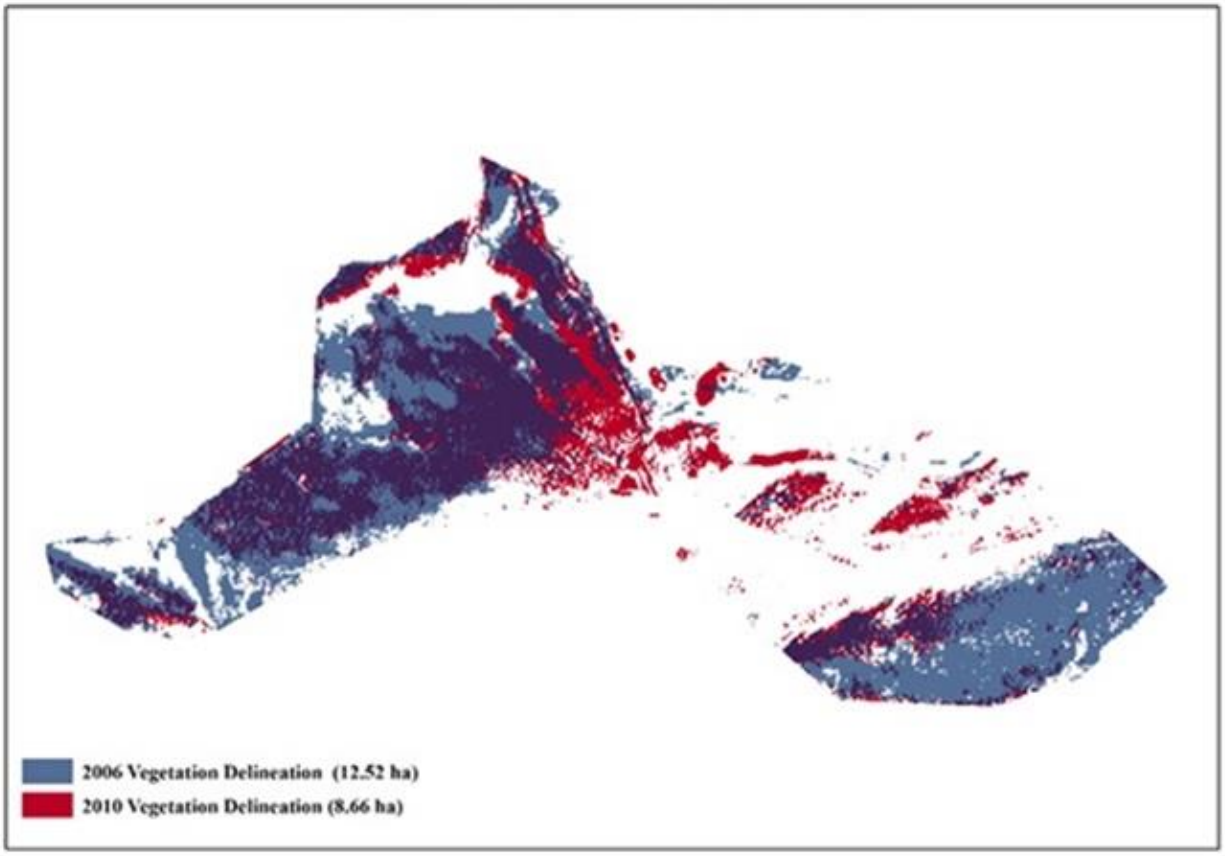

Figure 5: Southwestern willow flycatcher habitat at the Mesquite West survey site in Nevada. The difference in vegetation presence can be seen in the map to detect changes between 2006 Quickbird satellite imagery and 2010 CIR aerial photography. Purple denotes the overlapped areas, while blue represents vegetation present only in 2006 and red represents vegetation present only in 2010.

Visual interpretation of the change detection analysis confirms the effects of the 2005 flooding event. As evinced in the 2006 scene, the southeastern portion experienced scouring, and the Virgin River forged two new channels south and north of the bend observed in 2004. Other notable changes observed in 2006 included the shrinking area of the cattail marsh in the northern portion, where bare soil could be seen, and the drying of surface water adjacent to the golf course in the east. By 2010, the cattail marsh had disappeared, leaving a much larger barren area than in 2006. Vegetation growth occurred in some of the scoured section, including the northern bend of

the Virgin River as vegetation replaced stream water; however, more areas devoid of vegetation appeared in the west and in the central part, where surface water in 2006 vanished.

However, studying the maps for the visual interpretation of change offers only a qualitative assessment of the spatial distribution of change, not the magnitude of change. The graph illustrates a declining trend in the vegetation cover area (Fig. 6). In 2004, the vegetation cover measured 14.92 ha but shrank to 12.52 ha by 2006. This represented a 16% loss of vegetation cover. From 2006 to 2010, Mesquite West experienced an additional decrease of 30.8% (42% total change percent from 2004 to 2010) as the vegetation cover and available habitat for nesting, foraging, and patrol perches for the SWF dropped to 8.66 ha.

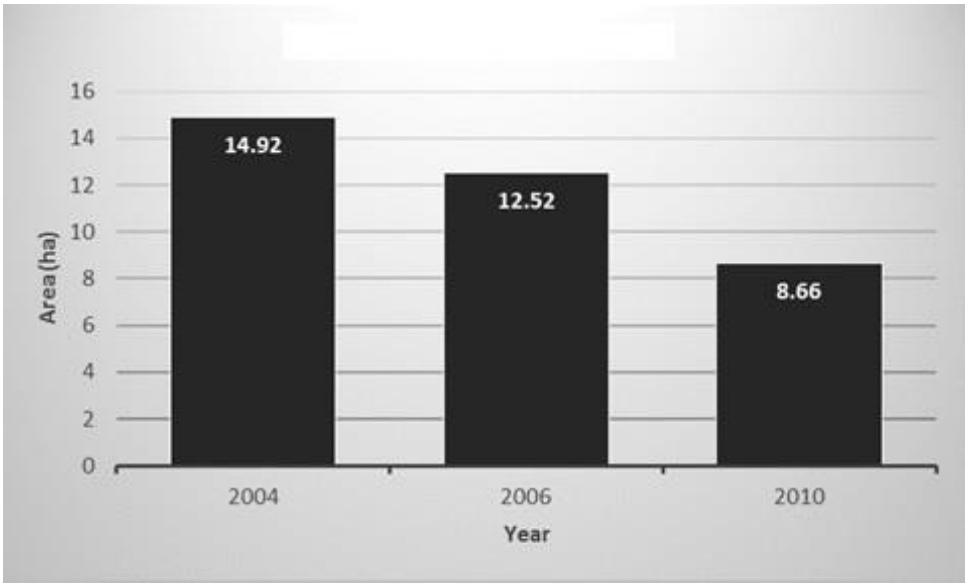

Figure 6: The bar chart graphs the areas of vegetation cover in 2004, 2006, and 2010. Note the decrease in the size of the columns through the years, indicating a declining trend in vegetation as calculated by the SPEAR Vegetation Delineation Tool.

The visual analyses of the three techniques for each year (Fig. 7 and Fig. 8) show marked differences in the vegetation extent, and these differences translated into a difference in the area

of SWF habitat (Fig. 9). Only the SPEARS Vegetation Delineation Tool reveals an obvious declining trend.

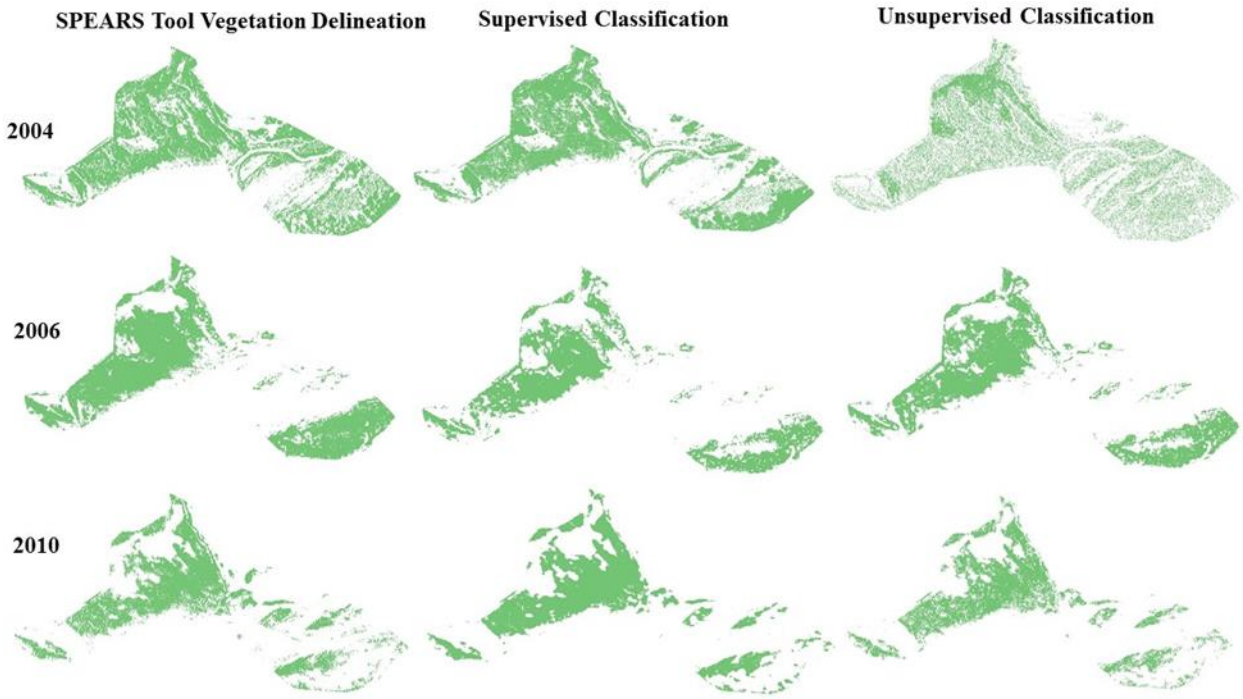

Figure 7: Graphic depicts the vegetation cover results from the three techniques (columns) for each year (rows).

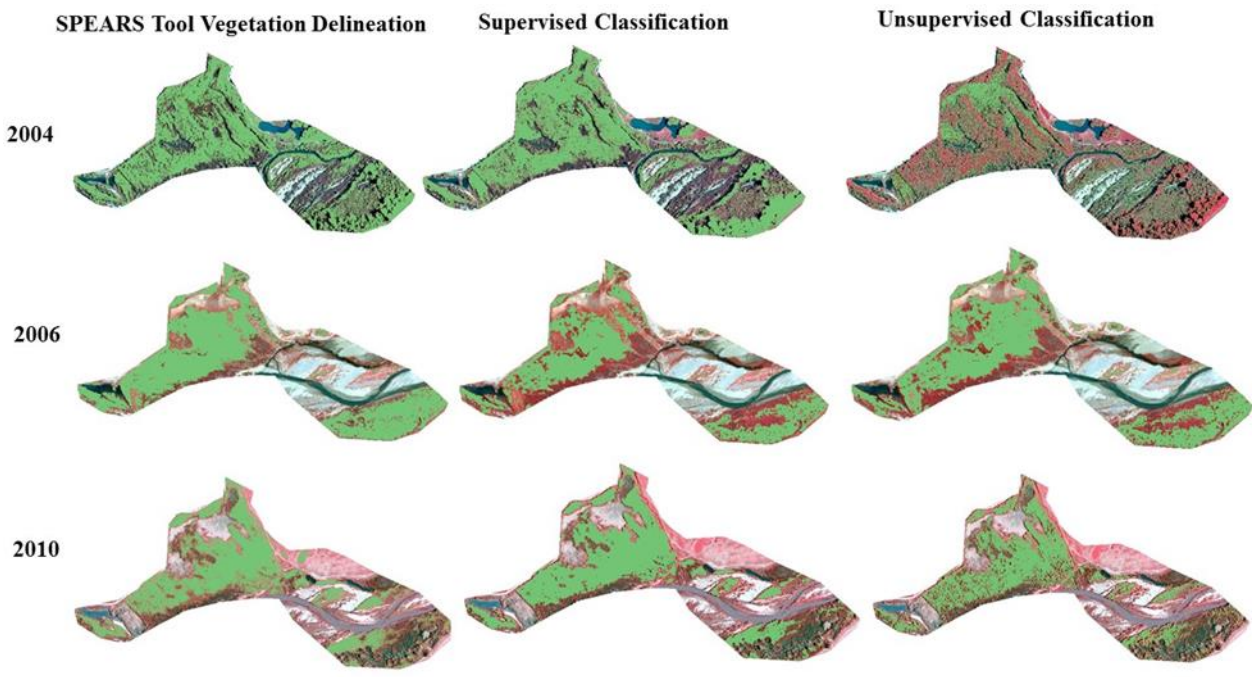

Figure 8: Graphic of the overlay mapping results show where the techniques have omission errors since the color-infrared displays vegetation as red, and the appearance of red indicates misclassification of pixels.

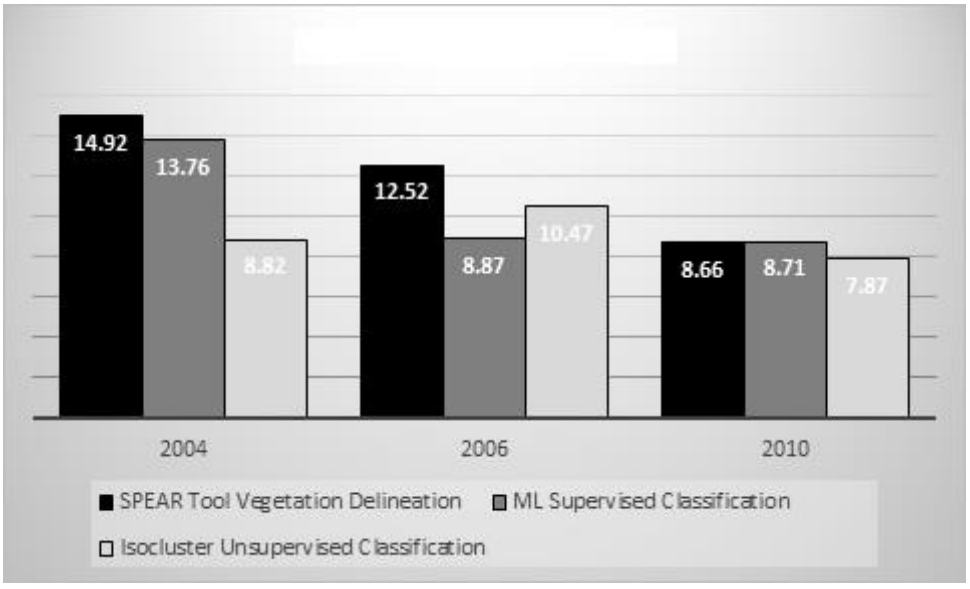

Figure 9: The bar chart for the three methods shows disagreement in the areal extent of vegetation cover as well as the trends over time.

For the 2004 CIR aerial photography, 2006 Quickbird satellite imagery, and 2010 CIR aerial photography, the accuracy assessment reveals the vegetation delineation technique had

higher overall, producer, and user accuracy and kappa coefficient than the supervised classification and unsupervised classification techniques (Table 1). This implies the SPEAR Vegetation Delineation Tool better captures live green vegetation than the traditional classification schemes for high resolution data and, being more accurate, may serve as a more appropriate method to perform change detection of riparian habitats or other heterogeneous landscapes.

Overall Accuracy	Vegetation Delineation	Supervised Classification	Unsupervised Classification
2004 CIR Aerial Photography	91%	80%	70%
2006 Quickbird Satellite Imagery	95%	82%	90%
2010 CIR Aerial Photography	94%	90%	92%
Kappa Coefficient	Vegetation Delineation	Supervised Classification	Unsupervised Classification
2004 CIR Aerial Photography	0.808	0.596	0.411
2006 Quickbird Satellite Imagery	0.9	0.642	0.8
2010 CIR Aerial Photography	0.857	0.766	0.822
Average Producer Accuracy	Vegetation Delineation	Supervised Classification	Unsupervised Classification
2004 CIR Aerial Photography	95.59%	80.45%	71.60%
2006 Quickbird Satellite Imagery	95%	82.35%	90.09%
2010 CIR Aerial Photography	91.45%	89.05%	92.50%
Average User Accuracy	Vegetation Delineation	Supervised Classification	Unsupervised Classification
2004 CIR Aerial Photography	91%	79.57%	71.10%
2006 Quickbird Satellite Imagery	95%	86.55%	90.20%
2010 CIR Aerial Photography	94.75%	86.35%	90.10%

Table 1: Confusion matrix results

Discussion

Riparian Habitat Changes and Conservation Implications

Any perturbation of the underlying processes will lead to modifications in the distribution, abundance, and composition of riparian habitats, and the results of this change detection analysis may help the scientific community better understand the vegetation responses

to shifts in the hydrological conditions. In this study, the most apparent change is the cattail marsh. SWF frequently nest near cattail marsh (Unitt 1998), build nests with the cattail tufts, and forage for insects within the habitat, above the canopy, above water, and along the patch edge (Sogge et al. 1997). In 2004, the cattail marsh functions as a surrogate indicator for the presence of hydric soils, i.e., waterlogged soils either saturated throughout the year by shallow standing water or by frequent or seasonal floods. The cattail marsh's disappearance in later years, therefore, alludes to the loss of the wetland hydrology that formed and maintained it, whether from abnormal precipitation fluctuations, water diversions, or groundwater withdrawals within the watershed. Determining causality would require climate, soil moisture, and groundwater data, not readily available for the study area. While beyond the scope of the present study, estimating soil moisture or evapotranspiration with satellite data could help determine the hydrologic regimes and productivity in each year and how they changed in time.

The apparent lack of surface water and soil moisture as time progressed did have a ripple effect throughout the site and the riparian-dependent SWF populations. The reduction of riparian area determined by this study puts migrants who search for suitable patches in shrinking or degrading habitat at greater mortality risks from competition, starvation, or predation, which may reduce or eliminate breeding opportunities and, in turn, cause declines in the local population (Marshall and Stoleson 2000). Since 2006, the number of females have steadily declined and, by 2013, only *one* female returned to the Mesquite West site (McLeod and Pellegrini 2013). The implications should be concerning since habitat suitability is ultimately measured by a steady or growing population (Sogge et al. 1997), and the question remains whether Mesquite West will continue to support riparian vegetation and/or SWF occupation.

Even if the site is declared unoccupied and unsuitable, it may still be considered potential habitat, especially if management practices will recreate suitability at Mesquite West (USFWS 2002). Suitable habitat matures or undergoes disturbance, thereby, becoming unsuitable; furthermore, all suitable habitat began as potential habitat (USFWS 2002). The dynamic nature of the riparian habitat means the habitat may cycle through periods of suitability and unsuitability, indicative of the resiliency inherent in these habitats. If managed appropriately, restoration of potential riparian habitats should assist the recovery of SWF populations.

The important findings of this study confirm a declining trend in riparian vegetation, and should inform and prompt managers into action. Nest productivity correlated with the irrigation flows (McLeod and Pellegrini 2013): higher during the years when the flows inundated the site (e.g., eleven nests produced eighteen fledges in 2008), low when the flows were disrupted (e.g., three of the thirteen nests each produced a single fledge in 2009), and increasing when flows returned (e.g., four of the fifteen nests produced seven fledges in 2010). Therefore, for the protection of the riparian habitat and survival of the critically endangered species, management efforts need to focus on maintaining irrigation flows to the site. Restoration along the Virgin River may also provide alternate nesting sites (McLeod and Pellegrini 2013). Management efforts should include uplands, headwaters, and tributaries because the health of the riparian ecosystem and nesting habitat depends on maintaining or restoring the whole landscape, not merely the nesting habitat, the smallest portion of the riparian ecosystem (USFWS 2002). Otherwise, the SWF may abandon the Mesquite West site like the populations did at eight territories along the Rio Grande in New Mexico (Marshall and Stoleson 2000).

These measures to document, quantify, and characterize the vegetation changes in the southwestern riparian ecosystems are imperative. As an oasis in semi-arid environments, the

riparian ecosystems in the southwestern United States constitute only 1% of the landscape (Nilsson and Svedmark 2002) but support 75% of the species during at least one phase of life (Kondolf et al. 1996), 60% of all vertebrate species, 70% of rare, threatened, and endangered species (Fischer et al. 2001; Poff et al. 2012), and a higher breeding diversity of birds than all other western habitats combined as well as highest noncolonial avian breeding densities in North America (Johnson et al. 1977). More than 50% of the songbirds, including the endangered SWF, depend on the critical habitat they provide (Knopf and Samson 1994; Hatten et al. 2010). Yet, dams, water diversions, and groundwater withdrawal have altered or eliminated the natural hydrology that forms and feeds the riparian ecosystems (Marshall and Stoleson 2000; Paradzick and Hatten 2004). Direct or indirect effects from other human activities such as urbanization, land development, mining, agriculture, and livestock grazing result in further declines in soil health, water quality, and biodiversity and alterations of hydrology (Patten 1998).

The smaller, more isolated patches of remaining habitat are more susceptible to stochastic events and may lack the resilience because of the reduction of biodiversity to recover, and endangered species, such as the SWF face greater risks of local extinction (Ellis et al. 2009) as their survival depends on the distribution and abundance of dense stands of riparian vegetation (Sogge et al. 1997; Hatten and Paradzick 2003; Durst et al. 2008; Paxton et al. 2007). A decline in SWF populations correlates with a decline in riparian habitat (Paxton et al. 2007); therefore, the SWF acts as prime indicator of ecosystem health (McCarter 1996). With habitat loss, degradation, and modification threatening the SWF (Sogge et al. 1997; Stoleson et al. 2000), conservation efforts have prioritized the maintenance of riparian environments and restoration of the functional qualities and values in the Southwest (Johnson et al. 1977; Knopf and Samson 1994; Finch 1999; Marshall and Stoleson 2000; Stromberg 2001; Hatten et al. 2010; and others).

These efforts will benefit at least 83 other species, including the endangered New Mexico jumping mouse (*Zapus hudsonius luteus*), the yellow billed cuckoo (*Coccyzus americanus*), Chiricahua leopard frog (*Lithobates chiricahuensis*), and the Least Bell's vireo (*Vireo bellii pusillus*). Humans benefit, too, from the water filtration, bank stabilization, nutrient cycling, sediment load reduction, scenic beauty, natural resources, and recreational opportunities (Patten 1998; Poff et al 2012). Through conservation management and restoration, we not only protect biodiversity and ecosystem health, services, functions, and benefits, but also riparian ecosystem resiliency. Resilience to climatic change and extremes associated with predicted climatic changes in the Southwest will be one of the most important factors in riparian habitat to persist.

Potentials and Limitations of the Hybrid Multi-Sensor Change Detection

SWF habitat fluctuates in response to water management and precipitation runoff (Paradzick and Hatten 2004). The spatially and temporally dynamic nature of the habitat should emphasize the need for more change detection research. One relevant research area is to correlate changes in flycatcher population to changes in the riparian habitat over time (Stoleson et al. 2000). In other sites where restoration has occurred, this study's hybrid change detection method also has the potential to monitor vegetation responses to the rejuvenation efforts and evaluate their effectiveness. Remote sensing change detection has emerged as an essential tool for the evaluation of conservation policies and management strategies (Corbane et al. 2015).

By 2027, 268 individual Committee on Earth Observation Satellites (CEOS) missions will be or are planned to be operating (Pettorelli et al. 2014). Recent and upcoming launches of new remote sensing platforms, such as Sentinel-2A (provides continuity for SPOT and Landsat missions), Amazönia-1, and EnMAP, compound the challenge of multi-sensor change detection. It would have been ideal to compare the same year with imagery acquired from different sensors

and determine how similarly the SPEAR Vegetation Delineation tool would have identified "live green vegetation." Still, the gravity of the matter is that the scientific community needs reliable and efficient means to monitor vegetation, especially riparian vegetation and its role in maintaining biodiversity. This paper achieves (1) its main goal to introduce a simple change detection technique that successfully handles the problem of data acquired from different high resolution sensors and (2) its objective to provide other remote sensing experts, ecologists, conservation biologists, policy makers, protected area managers, conservation consultants, and others a time-efficient, time-effective, sufficiently accurate and reliable means to monitor environmental changes. To that end, this study delivers a fast, simple, and effective change detection technique that meets the pressing needs of NGOs (Sader et al. 2001).

The main advantage of using this proposed method is its simplicity as compared to the recently developed, computationally complex multi-sensor change detection techniques (e.g., Habib et al. 2005; Alberga 2009; Pillai and Vatsavai 2013; Volpi et al. 2013; Pathak 2014), which require training not readily available or cost-prohibitive to many applied ecologists and conservation biologists. Even traditional same-sensor change detection techniques are computationally demanding, and the ones developed for multi-sensor data are limited in their utilization because of the complexity and ad-hoc target data fusion methods (Volpi et al. 2013). Application of the SPEAR Vegetation Delineation tool for change detection analysis also eliminates time-consuming pre-processing steps—such as co-registration, a process requiring precision (Lu et al. 2010), difficult to achieve particularly for high resolution data and shown to contribute errors to vegetation change measurements (Townshend et al. 1992; Duncan et al. 1993). It is an advantageous tool for the non-expert in remote sensing monitoring of landscapes and ecosystems for habitat loss, degradation, or fragmentation.

Despite high confidence in capabilities and assumptions of sufficient accuracy for the change detection analysis, its accuracy cannot be ascertained with any validity. A major limitation of this study revolves around the fact that ground truth data were lacking to undertake a more reliable statistical assessment. However, visual photointerpretation of high resolution data is regarded as reliable, repeatable, and accurate (Tiner 1996; Fensham and Fairfax 2002; Valta-Hulkkonen et al. 2005; Ihse 2007; Yang 2007; Morgan et al. 2010 and others), given the high image quality, fine details, and geometrically and visually true representations of a scene. False color aids in the photointerpretation—e.g., green and light pink indicate low plant density or dying/dead vegetation (Statewide Mapping Advisory Committee 2001). The ability to discern healthy vegetation from the dying/dead facilitates the visual interpretation of the NDVI threshold; although, accuracy will depend on the skill, visual acuity, and *a priori* knowledge of the analyst.

With the results from the confusion (i.e., error) matrix, the core of quantitative accuracy assessments in remote sensing (Foody 2002), the thematic maps derived from the supervised and unsupervised classifications complements the basic visual analysis of accuracy and suggests the SPEARS Vegetation Delineation tool outperforms the traditional classification methods. Compared to the SPEARS vegetation delineation, the supervised and unsupervised classification scheme underestimated the extent of vegetation, which the overlay maps of the vegetation feature classes and false-color imagery of the site substantiate by how much more red (indicative of healthy vegetation) is revealed in the supervised and unsupervised classification maps. The supervised classification appears to have more accurate results with the CIR aerial photography than the Quickbird satellite imagery, and the unsupervised classification has evident commission and omission errors in 2004. Any errors in the classified images would amplify errors in the

post-classification change detection and indicate false changes (Singh 1989). No doubt the spectral and spatial heterogeneity of high resolution imagery data for riparian zones hindered the use and accuracy of the conventional pixel-based image analyses, which assume homogenous features (Johansen et al. 2010). However, this assumption does not impinge upon the SPEARS tool operation.

The SPEARS Vegetation Delineation tool, though, encompasses more than subjective visual interpretation, the standard for analyzing aerial photography (e.g., Heiskanen et al. 2008; Ihse 2008). Another advantage of its use lies in the semi-automated process for the NDVI calculation. Because of the strong correlations with biophysical properties of plants, NDVI serves as the principle parameter of plant condition, biomass, vegetation cover, leaf area index (LAI), and the fraction of photosynthetically active radiation (FAPAR) as well as the most influential covariate of SWF breeding habitat models (Hatten et al. 2010). Its application to monitor temporal changes in vegetation has been well documented (Carlson and Ripley 1997; Lyon et al. 1998; Nagler et al. 2001; Kerr and Ostrovksy 2003; Lunetta et al. 2002; Lunetta et al. 2006; Manciono et al. 2014; Ghandi et al. 2015; and others). Hatten et al. (2003) suggested manipulating NDVI values to reflect proposed changes in a simulation model or to run models before and after activities that influence habitat characteristics as a way to understand changes in quality and abundance of breeding habitats. Two studies coupled habitat suitability variables and NDVI probability classes to monitor changes in predicted SWF habitat (Paradzick and Hatten 2004; Paxton et al. 2007). For change detection analyses, NDVI performs better than other vegetation indices as a predictor of vegetation cover (Nagler et al. 2001), and the NDVI difference image aligned more with the visual interpretation of change and field work (Lyon et al. 1998), indicative of its accuracy. Because NDVI more accurately identified the spectral

signature of "live green vegetation" than the pixel-based classifications, the change detection analysis in this study would have higher accuracy than a post-classification change detection. Another advantage in change detection, NDVI can reduce multiplicative noise, such as illumination differences, shadow, atmospheric attenuation, some topographic variations, within multiple bands (Huete et al. 2002; Chen et al. 2005).

Typically after image NDVI transformation, a pixel-based or object-based classification scheme is applied to complete the multi-sensor change detection procedure (e.g., Millward, Piwowar, and Howarth 2006; Forkuo and Frimpong 2012; Pathak 2014; Mandanici and Bitelli 2015). Even with the more robust object-oriented classification, misregistration and misclassification can transpire and introduce errors. Future directions do include, however, comparing the hybrid change detection technique in this study to a post-classification with object-oriented classification of the "live green vegetation." Importing the NDVI-derived vegetation mask into ArcMap to easily calculate the area of vegetative cover also will reduce any additional potential errors and significantly cut processing time down to 5 minutes or less. Overall, the entire change detection procedure, from pre-processing to map results, takes less than an hour. No other method can produce results as quickly. Still, an interesting experiment may be to perform a more rigorous atmospheric correction to see if the SPEAR Vegetation Delineation Tool would produce different results.

While NDVI promotes the advantage of reducing the noise effect on non-uniform illumination, such as from aspect, which improves the comparability of the index across the image, specific deficiencies associated with NDVI may need corrected (Jones and Vaughn 2010). For example, the Green NDVI (GNDVI) better detects chlorophyll as it increases over a much wider range than NDVI and, therefore, improves estimations at higher LAIs; the Soil-

Adjusted VI (SAVI) corrects for soil reflectances (Jones and Vaugh 2010). In this study, surface soil and water reflectances may have affected the NDVI values of the canopy and the apparent "greenness" of the vegetation in the images. Perhaps modification of the index or combination with another index, such as the Normalized Difference Water Index (NDWI) used for wetland mapping and change detection in a recent study (Kavyashree and Ramesh 2016), would refine the delineation and enhance the change detection performance. Other criticisms of NDVI hinge upon its inability to detect relative contributions of grass and tree canopy within a measure (Parrini, Macindoe, and Erasmus 2013), which confounded the delineation of the riparian vegetation in the current study. This is one disadvantage with using NDVI rather than a classification scheme because of the limitation in detecting vegetation by density, not by type.

Overall, the creative use of ENVI's SPEAR Vegetation Delineation tool and ArcGIS for change detection analysis performed better than expected for high resolution imagery data derived from multiple sensors. CIR aerial photography and Quickbird satellite imagery, however, may be too cost prohibitive or unavailable for some studies. Landsat offers cost-free, publicly available imagery with large archive, continuous record, and future repeatability, but the applicability of the proposed method to handle moderate resolution data with high resolution data or other resolution data acquired from different sensors as well requires further experimentation.

Researchers may also confront the inaccessibility of the software; however, analysts may be able to use open source products, such as RStudio or QGIS, and programming languages to calculate and manipulate the NDVI ranges for vegetation delineation, mapping, and monitoring, albeit with more effort and time involved. But, more collaborations between remote sensing

experts and ecologists may help bridge the data, software, and knowledge gaps as well as help purse common environmental research questions (Pettorelli, Safi, and Turner 2014).

Conclusively, this study links remote sensing with ecological research and conservation application. With the current rate of extinction 100-1,000 times greater than pre-human rates (Pimm et al. 1995) and landcover change as one of the main driving variables that impacts biodiversity (Chapin et al. 2000), the ability to monitor landcover changes affecting the basic processes within the environment is fundamental to conserving biodiversity and to designing appropriate management strategies (Pettorelli, Safi, and Turner 2014). Remote sensing is a powerful, vital tool for these tasks as well as a prominent research topic (Corbane et al. 2015). More integration between ecological research and remote sensing science and knowledge transfer is required to keep pace with the accelerated changes occurring around the world (Aplin 2005; Pettorelli, Safi, and Turner 2014; Pettorelli et al. 2014; Corbane et al. 2015; Willis 2016).

Conclusion

In summary, a change detection method developed originally for CIR aerial photography was applied successfully to high-resolution multi-sensor data. ENVI's SPEARS Vegetation Delineation tool delimited the riparian vegetation cover of the Mesquite West site in Nevada, and the changes in the SWF habitat from 2004, 2006, and 2010 were mapped in ArcGIS. The study represents the first attempt to quantify the SWF habitat changes at the Mesquite Site, which revealed a declining trend. The conclusion of this study recommends the use of the method described to monitor changes in habitat area or quality in all SWF territories. For other species dependent on the extent and quality of vegetation cover, the employment of this method would be beneficial as well. The ease of implementation and the rapid, sufficiently accurate results

provide high value to the hybrid change detection method, especially for a discipline where time is of the essence.

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We report no potential sources of conflict of interest, such as patent or stock ownership, membership of a company board of directors, membership of an advisory board or committee for a company, or consultancy for or receipt of speaker's fees from a company.

Data Accessibility

Due to the sensitive nature of information on the locations of endangered species, the data for this research cannot be archived publicly. Requests for the data may be directed to the Bureau of Land Management and the Bureau of Reclamation. The agreement with these agencies to use their data for research prohibits sharing in any manner.

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CHAPTER FIVE

CONCLUSION

Geography should stand at the forefront of the biodiversity crisis, armed with the appropriate tools and knowledge to combat the problems. The abiotic and biotic components of the physical environment are interrelated and interdependent. Therefore, we cannot conserve the biotic without maintaining or restoring the abiotic factors that contribute to ecosystem functioning and life-provision for organisms. Within the field of physical geography, we gain that understanding of how the Earth works. Through human geography, we study the human-environment relationship, how human activity affects landscapes and how landscapes influence human culture and activity. To conserve biodiversity, we will need to borrow from both sub-disciplines because human activity drives biodiversity loss and biodiversity conservation needs human participation. Geographic research can more effectively answer the three key questions: (1) "What is where?" to explain biodiversity patterns and to identify ecological variables and priority areas; (2) "What is changing where?" to alert managers about conditions, to inform actions, and to investigate effectiveness of strategies; and (3) "What will be where?" to predict patterns under specific, future conditions (Bregt et al. 2002).

The research articles introduced in this dissertation undertook the first two questions. By answering those questions with the use of ecological data and geospatial technologies, the research may contribute to current conservation programs: (1) the habitat suitability modeling identified priority areas where potential reintroduction of red wolves into the Daniel Boone National Forest may occur and (2) the change detection analyses showed where and how much change (loss) had occurred in the endangered southwestern willow flycatchers' critical riparian habitat in Mesquite, Nevada. Managers and decision-makers within the U.S. Fish & Wildlife

Service and the Bureau of Land Management, respectively, can use this pertinent information to advance their initiatives. Timely dissemination of the information enables the effective development and implementation of solutions (Buchanan et al. 2015).

These findings are important, not only for adding to the body of knowledge about specific habitat suitability or changes, but also because of the implications for practice. Restoration of wildlife first requires an understanding of the habitat criteria that shape the distribution, abundance, and persistence of species, and we cannot stem habitat loss without first monitoring and documenting habitat changes and the factors influencing the changes (Morrison 2013; Szantoi et al. 2016).

To increase the probability of rewilding success, we must release the species into core historic range and into high quality habitat (Morrison 2013). The habitat suitability modeling described in this dissertation provides the means to locate high quality habitat for any species, especially extirpated species, rare and endangered species, species less commonly studied, and for those lacking presence/absence data. Applying the results of the specific modeling efforts, we can return the red wolf to its historic range, where it can establish a home range in Kentucky. Other restoration sites in the remaining large Southeastern U.S. forests do still need to be located as well. Re-establishment of an apex predator and/or a keystone species, such as the red wolf, also introduces the possibility of increased genetic variation and evolutionary potential as well as improvement on biodiversity and ecosystem health (Noss 2001).

Since habitat loss drives the biodiversity crisis and given the magnitude of the threat, we need strategies to easily and quickly monitor habitats (Szantoi et al. 2016). Information on the amount of habitat loss enables predictions of biodiversity loss as well as can indicate an immediate need for conservation action to preserve biodiversity (Francis and Goodman 2010).

At the Mesquite West site, the declines in riparian habitat and the southwestern willow flycatcher population do evince the need for restoration, and hydrological flows will need to be managed to satisfy political and societal needs and to produce desired ecological response. The site should be preserved for breeding populations, but at the very least, as a suitable habitat for dispersal to maintain viable populations. A crucial next step in research is to do a wide-range southwestern willow flycatcher habitat analysis employing the rapid change detection method presented in the last two articles. With those results, managers can perform a habitat-based population viability analysis and assess the potential impact on the species.

This dissertation research provides widely *applicable, practical, and employable* geospatial models to perform habitat assessments for biodiversity conservation. Considering the expertise problems of using GIS and RS for ecological modeling, the easy-to-implement techniques provided for the conservation community to perform habitat suitability and change detection analyses fills a pressing research gap (Bregt et al. 2002; Busby 2002; Buchanan et al. 2015; Palumbo et al. 2016). Tailoring the dissertation research to management needs is another significant step in bridging the gap between RS/GIS specialists, ecology, and the conservation community (Pettorelli and O'Brien 2014; Corbane et al. 2015; Mairota et al. 2015). The lack of integrated knowledge of GIS, RS, and ecological modeling, trained personnel, expertise, cross-disciplinary work, and practical know-how compound the challenges faced in conservation science (Bregt et al. 2002). This necessitates the development and sharing of more "user-friendly" techniques for common conservation research topics, such as habitat suitability modeling and change detection analysis. In lieu of offering training opportunities to improve skills (Palumbo et al. 2016), offering alternatives to the more sophisticated techniques makes practical sense. Training takes time and money, and, as long as the analyst has basic RS/GIS

skills and understanding of RS data, the more simplistic models serve as more efficient means to rapidly procure answers and expedite solutions. Successful environmental management depends on timely, accurate, and detailed information on ecosystem variables and changes (Skidmore 2002).

While the studies deliver "user-friendly" techniques to a broad audience, modeling limitations do exist. Rigorous accuracy assessments could not be performed. In all the research, the lack of appropriate data (e.g., absence/presence data and ground control points to determine the number of correctly classified areas) prohibited verification of results. Data inadequacies represent the bane of conservation research (Bregt et al. 2002; Richardson and Whittaker 2010; Whittaker et al. 2015). However, the lack of rigorous accuracy assessments should not hinder the usefulness or denounce the reliability of the models. The models still provide new insights and prompt further investigation. We have uncertainty inherent in every aspect in biodiversity conservation: (i) from our level of understanding of the highly complex, dynamic, unpredictable nature of ecosystems (ii) through data shortages/quality and model development (iii) to decision-making, policy-planning, and implementation mostly done by those with little scientific training, and (iv) even in the effectiveness of the actions (Francis and Goodman 2010). In the end, we rely on professional judgement.

Future directions of the research include exploring ways to quantify and assess accuracy. The wolf habitat suitability model may be tested in gray wolf-occupied territories, but this depends on the ability to obtain presence/absence data. For endangered and threatened species, data confidentiality on locations restricts public availability. The southwestern willow flycatcher data used in the two studies, for example, was obtained via formal request, written proposal, and a strict agreement on the conditions of use. While bird presence, nests, and territory centers were

georeferenced and known from these data, no ground control points for ground truthing had been taken for habitat perimeters during any year. Perimeter ground control point data would assist in verifying wetland delineation results (*see* e.g., Barrette et al. 2000), and scientists planning wetland change detection analyses should consider collecting these data at the start and end of a study period. If this task has been undertaken for the Mesquite West site, accuracy of the vegetation delineation could have been assessed with more rigor than visual interpretation. Accuracy of the change detection rested on the accuracy of the vegetation delineation.

Another direction for future research is testing the techniques within new contexts, whether adopting a different critically endangered species, such as the Florida panther (*Puma concolour coryi*), or different ecosystem type, such as a tropical rainforest or even coral reefs. The rapid change detection method can answer one of the top conservation questions and provide information on habitat extent and change in extent, especially on wetlands which are of critical interest (Buchanan et al. 2015). To meet other conservation needs that are integral to stalling biodiversity loss, the habitat suitability model may be modified to see if it can identify hot spots (Corbane et al. 2015). By applying the methods beyond the limited scope of the dissertation research, we demonstrate their wide-ranging utility to conservation science.

Despite the appeal of the models' simplicity, the incorporation of complexity, such as modeling uncertainty or valuating ecosystem services, would greatly enhance the significance of the research. The importance of maintaining or restoring biodiversity within the studies hinges on the intrinsic values of ecosystem structure and function. Policy-makers, stakeholders, and community members may require economic incentives. Therefore, the addition of the valuation of ecosystem services would make a more compelling argument for biodiversity conservation, and management plans can be implemented in a way that balances the different interests at

different scales (Hein et al. 2006). Quite a bit of research on the valuation ecosystem services and capital exists, but difficulty still surrounds quantifying the existence values (van Wilgen et al. 1996). Hein et al. (2006) provides a framework that to evaluate ecosystems based on three types of services—(1) *regulation* (e.g., regulation of climate, water, pollution, pests, and pollination; (2) *production* (e.g., genetic resources, food, fuel, and medicines) and (3) *cultural* (e.g., cultural, religious, or historical heritage and recreational and tourism opportunities) and four types of values—(1) *direct use*, (2) *indirect use*, (3) *option*, and (4) *non-use* values. With this framework, we can estimate the costs and benefits with biodiversity and nature conservation (non-use value). The Nature Capital Project spearheaded by the Stanford University in partnership World Wildlife Fund provides a tool, InVest, to calculate the costs and benefits of different ecosystems and estimate the benefits from ecosystem restoration. Since ecosystem services vary by spatial scale, geographic research for biodiversity conservation should devote efforts to the valuation of ecosystem services.

Geographic research will serve as the solution for many conservation challenges. RS and GIS specialists can lead the development of more highly accessible and applicable methods to help us understand biodiversity patterns and distributions and the changes in biodiversity, as well as the processes that maintain ecosystem integrity. Being a part of the solution also means integrating RS and GIS into conservation programs, collaborating with other disciplines, mobilizing information between interest groups, and incorporating the social perspective into research (Brown 1999; Skidmore 2002; Bregt et al. 2002; Draper et al. 2003; Francis and Goodman 2010; Pettorelli and O'Brien 2014; Pettorelli et al. 2014; Buchanan et al. 2015; Corbane et al. 2015; Mairota et al. 2015). Geography should concentrate more intensely on biodiversity conservation, which will effectively synthesize the dichotomy between physical and

human geography. This will not only help geography survive, but ecosystem health and our well-being may hinge upon this capacity building within geography and geographic technologies for all conservation matters.

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