Geospatial Analysis of Forest Fragmentation and Connectivity in Virginia

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SCIENTIFIC ABSTRACT

In areas where human activities have led to deforestation and forest degradation, forest fragmentation occurs at high rates, affecting biodiversity and wildlife populations. In Virginia, where human population growth rate is estimated at approximately 5% annually, forest fragmentation is considered as the main cause of local population decline of forest birds such as the Ovenbird and Wood Thrush, as a result of urbanization. Analysis of the extent to which forest fragmentation and connectivity have occurred in Virginia and the corresponding changes associated with these processes, is relevant for conserving forest habitats and the biodiversity they support. With increasing availability of satellite imagery at multiple scales and finer resolutions, it has become possible to understand effects of comparing fragmentation metrics acquired from coarse resolution images that are relatively easy to acquire because of their economic costs, and metrics obtained from finer resolution imagery that are more expensive and therefore, span only small areas. This study investigates the extent to which forests in Virginia have become fragmented or connected between 2001 and 2011 using satellite imagery of varying spatial resolutions, and how forest fragmentation and connectivity rates affect populations of Wood Thrush and Ovenbirds. Management policies that could curb forest fragmentation and help in species conservation are also investigated.

This research combines use of ArcGIS, FRAGSTATS and R software programs to provide accurate estimates of forest fragmentation in Virginia. In this research, a Markov Chain Monte Carlo model is created that looks at effects of differing variables such as slope and fragmentation on bird populations. This model can be replicated and improved by other researchers for specific bird populations and differing study areas. This research applies geospatial analysis to evaluate efficiency of riparian buffers in Virginia. This approach identified how forest habitat in Virginia can be improved.

Results of this research indicate increasing rates of forest fragmentation in Virginia between 2001 and 2011 and the negative effect of forest fragmentation on Wood Thrush and Ovenbird populations. This research also identifies how human population density negatively affects the size of ecological patches and the role of riparian buffers in serving effectively as corridors to link ecological patches. Results of this research can be generalized to other states for forest conservation and management.

Geospatial Analysis of Forest Fragmentation and Connectivity in Virginia

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GENERAL ABSTRACT

This research evaluated the extent to which forests in Virginia have either become fragmented (disconnected) and/or connected over a ten year time period. The study analyzed the accuracy of forest fragmentation analysis depending on the spatial resolution of the satellite imagery used. This analysis highlights the importance of using appropriate satellite images for forest fragmentation analysis. Secondly, this research focused on building a model to identify the significance of factors such as slope, physiographic region and forest types on Virginia's populations of Wood Thrush and Ovenbird. This assessment identified the difference in effects of variables on bird populations depending on the scale at which the analysis is carried out. Third and final analysis combined the first two assessments to determine how management policies can be used to mitigate negative effects of forest fragmentation and protect biodiversity. The research results highlight increasing forest fragmentation trends in Virginia between 2001 and 2011 and the negative impacts of this trend on Wood Thrush and Ovenbird species. The results also demonstrate the effectiveness of riparian buffers as corridors.

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Chapter 1: INTRODUCTION, STATEMENT OF PURPOSE AND LITERATURE REVIEW

1.1 Introduction

Numerous processes, including rapid population increase, pollution, and high urbanization rates, drive global change, creating impacts that threaten conservation of natural resources, and reduce biodiversity at global, regional, and local scales. In many biological systems, such processes generate problems such as species invasion, and anthropogenic habitat alteration; all identified as significant environmental stressors (Strayer and Dungeon, 2010). Destruction of habitats typically leads to habitat fragmentation, a process that initiates long-term changes in the structure and function of the resulting patches (Haddad et al., 2015).

Impacts of anthropogenic processes on diversity and distribution of plants and animals have a long history, spanning thousands of years, including agricultural practices, accelerated urban development, and road and dam construction (Solbrig and Solbrig, 1994). Human-induced changes have proven to reduce and simplify biological diversity, at both the species and genetic level. ***Forest fragmentation*** - the division of forest habitats into smaller, less connected (disconnected) patches, results largely from human-caused processes. ***Connectivity*** describes the degree to which individual habitat patches in a landscape are connected. ***Corridors*** are defined broadly as permanent or temporary connections or linkages between core habitat areas (Bond, 2003). Brown and Kodric-Brown (1977) defined corridors as long, thin, strips of habitat that connect otherwise isolated habitat patches, and have the capacity to increase local species abundance and diversity by joining isolated populations. From that earlier definition, recent publications have highlighted that corridors do not necessarily need to be similar to the habitats they connect (Beier and Loe, 1992; Perault and Lomolino, 2000). Corridors could be roads or grassland habitats that connect forest habitats. ***Human-constructed infrastructure***, such as roads, can form important corridors for biological populations but also can contribute to isolating populations in instances where large highways and other infrastructure divide habitats. Thus, it is common to identify intact habitat conditions at one end of a continuum, with a completely developed urban area on the other, and isolated patches in the middle of the continuum.

Remote sensing is a useful tool to compare changes in forest habitats over a time gradient to identify human-induced changes and other natural processes that cause fragmentation and create corridors. Ultimately, such processes affect local wildlife populations. Causes of forest fragmentation (e.g. roads, agriculture, fire, human settlements) can be mapped to identify their specific impacts on intact habitats as well as upon disturbed habitats. Landscape changes can be traced through time and compared with populations of diverse biological species at corresponding times to assess impacts of anthropogenic factors.

Remote sensing has the capacity to provide systematic temporal observation data of the Earth at scales from local to global (Wang et al., 2010). Currently, the status of remote sensing systems allows for acquisition of fine to coarse spatial resolution, hyperspectral, and thermal imagery. For instance, while Landsat imagery provides medium to coarse resolution (approximately 30m), base map imagery of states such as Virginia, are of relatively finer resolution. Further, imagery at even finer scales, often less than 5 meters, can be acquired using Unmanned Aerial Systems (UAS). High-resolution imagery of small areas acquired using UAS, could be used in coordination with Landsat imagery and aerial photography at relatively coarser resolutions, to better delineate landscape patches and provide more detailed analyses of the extent of forest fragmentation and connectivity. Results from such analyses, as well as knowledge of critical forest species interactions, will be useful in the formation of more effective forest management and biodiversity conservation policies.

A wide range of scientific literature supports the view that anthropogenic activities that fragment natural forest habitats is an important cause of the decline and loss of species in any given environment (Didham et al., 2012; Fahrig, 2013). As a result of constantly changing landscapes and increasing urbanization rates, Virginia presents an opportunity to study effects of forest habitat fragmentation, the significance of habitat corridors, and effects of anthropogenic activities on wildlife populations.

Studies by the Virginia Department of Forestry (VDOF) Forestland Conservation Program highlighted the fact that family-owned forestland is most susceptible to fragmentation and land cover changes, especially at the point of intergenerational transfer of family property (VDOF, 2016). In Virginia, 10.6 million acres of private forestland, representing 41% of all private forestland, are owned by people over the age of 65 years (VDOF, 2016). This ownership pattern means that a significant fraction of forestland in Virginia is at risk of conversion, or has already undergone changes over the years. Thus, Virginia's historical context offers a setting for investigating forest habitat fragmentation and its impacts.

1.2 Statement of Purpose, Significance and Dissertation Outline

1.2.1 Statement of Purpose and Significance of Dissertation

Forest habitat fragmentation is caused mostly by increasing changes to the landscape via human activities and poses a significant threat to biodiversity conservation by decreasing sizes of forest habitat patches and increasing their isolation by subdividing remaining habitats (Hanski, 1999; Brooks et al., 2002; Fahrig, 2003). In this context, successful dispersal rates of organisms decrease significantly, and fewer individual populations are able to establish and persist within smaller and isolated patches (Harrison and Bruna, 1999). Forest fragmentation therefore has negative impacts on local, regional and global species abundance (Harrison and Bruna, 1999) as a result of increases in vulnerability of biotic species to physical and biological threats (Franklin et al., 2002; Wethered and Lawes, 2003). With tropical forests becoming increasingly fragmented as a result of intensification of anthropogenic activities, examining the extent of forest fragmentation is imperative for maintaining populations of biological species.

There are numerous biological species classified as habitat specialists inherently vulnerable to forest fragmentation (Cushman, 2006). The "habitat heterogeneity hypothesis" states that increases in the number of habitats and/or increases in their structural complexity leads to increased species diversity (MacArthur and Wilson, 1967; Connor and McCoy, 1979). This implies that increasing the number of habitats in a given region by creation of smaller patches will result in a wider variety of species via increased number of niches (Tews et al., 2004). However, some species prefer habitats of differing sizes for diverse functions during different life stages. Hence, there is a need to analyze habitat sizes, depending on characteristics and preferences of species present in the region. Although the habitat heterogeneity hypothesis partially explains the biological diversity of an area, it requires a habitat scale factor to ensure thriving species populations.

Although forest cover reduction significantly reduces the number of species in a given habitat as a result of a reduction in total available habitat for species, species that have broad habitat tolerance and are highly mobile can thrive in fragmented habitats, depending on characteristics of the fragmented patches (Nupp and Swihart, 1996). For instance, the white-tailed deer, *Odocoileus virginianus*, has been identified as a successful inhabitant of forests fragmented by agricultural lands due to differences in overstory cover. Other species, such as the eastern fox squirrel and the white-footed mouse, have been identified as possessing attributes that enable them to thrive in such habitats. Despite the success of such species in fragmented habitats, the overall impact of fragmentation is still unknown (Nupp and Swihart, 1996). The extent of fragmentation significantly affects the severity of edge and patch size effects (Sallabanks et al., 2000). Thus, it is imperative to understand the influence of patch sizes, edge proportion and patch isolation (connectivity) on species richness and abundance, given the characteristics and preferences of the species present.

In any given landscape, connectivity determines those portions of the total habitat accessible for an organism located in a particular point in the landscape (Saura et al., 2011). Connectivity also takes into consideration the ability of species to move from one habitat area to another through non-habitat areas (Tischendorf and Fahrig, 2000). Therefore, connectivity is not only important within individual habitat units but also aids in movement of species between areas, linking different habitats (Manning et al., 2009; Saura et al., 2011).

Mladenoff et al. (1993) reported that anthropogenic activities such as timber harvesting altered the natural landscape differently than natural disturbances. In 1998, Tinker et al. concluded that patches resulting from human disturbances, specifically timber harvesting, had more edge effects compared to patches that result from natural disturbances. In characterizing a range of patches, Hudak et al. (2007) compared patch characteristics of forest stands that replaced harvested trees to those that regrew after fire disturbances. They concluded that timber harvesting created smaller fragments with longer inter-patch disturbances and less interior habitat, compared to naturally disturbed patches. Throughout Virginia, logging has gained prominence in recent years as more forested areas are cleared for residential construction. A diverse literature supports the conclusion that timber harvesting tends to remove a larger amount of biomass from the forest (Tinker et al., 1998), and fragments the landscape differently, compared to more natural deforestation mechanisms, such as fire. Therefore, it is important to identify the degree of forest fragmentation resulting from human activities such as logging, and its consequent impact on land cover and biodiversity in Virginia.

- ***Key species for research***

The Wood Thrush, *Hylocichla mustelina,* is a forest bird species known to be very sensitive to forest habitat fragmentation (DeGraaf and Rappole, 1995; Weinberg and Roth, 1998; Robbins et al. 1989). Breeding sites of the Wood Thrush include mixed deciduous forests and damp woodlands with shrub sub-canopy layers. It is a ground forest bird that feeds on macroinvertebrates such as beetles and ants that live in leaf litter and on forest floors. Further, Wood Thrush nests are made of herbaceous stems and leaves (DeGraaf and Rappole, 1995; Roth et al., 1996). As this bird species thrives best in the core of forested areas, Wood Thrushes are impacted negatively by forest disturbances and do best when distant from human settlements (Roth et al., 1996). With low numbers recorded near paved roads and powerline edges, the nature of corridors that link two forest habitat patches is important for this species. Hence, it is important to study specifics of the impact of forest fragmentation and connectivity on Wood Thrushes in Virginia. For the kind of research questions targeted in this study, Wood Thrush is an excellent study species.

The Ovenbird, *Seiurus aurocapilla*, is another fragmentation-sensitive neotropical migrant wood warbler (Morimoto at al., 2012), named for its unique domed, oven-shaped, nest. Ovenbirds require large areas of undisturbed forest for successful breeding as they nest on the ground and smaller forest patches make them especially susceptible to brood parasites, specifically the Brown-Headed Cowbird (Robinson, 1988). Thus, knowledge of the impact of forest fragmentation and connectivity on this species in Virginia, which forms a significant section of its native habitat, will be useful in determining how the species can be protected.

Robin et al., (2003) reported that the populations of forest birds have decreased by about 30% as a result of forest fragmentation leading to isolation. Particularly for forest specialists such as the Ovenbird and Wood thrush, forest patch characteristics such as area and distance between forest patches, have significant effect on their populations (Robinson et al., 2000). It is important to note however, that forest patch characteristics do not always have the same effect. For instance, in a study by Estrada et al. (1997), forest patch area had a negative effect on the populations of birds in Mexico. Given this varying effects of forest fragmentation on birds, it is important to understand specific impacts of forest fragmentation on particular species before decisions are made. Such knowledge forms a foundation for defining effective forest management and biodiversity conservation policies.

- ***Remote Sensing***

Remote sensing technologies present a reliable technical approach to monitor landscape changes and analyze their causes and impacts (Chen, 2002; Wu and Murray, 2003). Landsat provides moderately coarse resolution imagery of the Earth, providing the longest, continuous global record of the Earth surface. Beginning in the early 1970s, Landsat has consistently archived images of Earth, enabling research scientists to assess changes in Earth's landscape. Aerial photographs form the basis of remote sensing applications, serving as aids for visual documentation of waterbodies, roads, vegetation, and geomorphic changes. In Virginia, the Virginia Base Mapping Program (VBMP) acquires statewide aerial photography of medium resolution, to promote effective and economically efficient development and sharing of geospatial resources across the state. Controlled from the ground, Unmanned Aerial Systems (UAS) provide high-resolution imagery of the Earth's surface, enabling identification of land parcel edges and structures such as roads, compared to Landsat imagery.

Thus, analysis of Virginia's landscapes using a historical archive of Landsat data presents a unique opportunity for identifying the temporal trajectory of landscape patterns, thereby providing an ability to quantify the impact of forest fragmentation and connectivity. The spatial resolution of Landsat and other satellite imagery is ideal for identification and correlation of patch metrics of larger forest patches with biodiversity and species abundance at coarse landscape scales (Newton et al. 2009). However, accurate spatial quantification of fine-scale features such as the edges of roads and gaps of small sizes are not resolvable at the resolution of such imagery (Fujita et al. 2003; Turner et al. 2001). Fine-scale correlations with biodiversity populations are crucial for assessing specific land-use intensities in forested areas since forest disturbances, including logging, occur at a fine scale within continuous areas or large patches of forest (Getzin et al., 2011).

Hence, specific objectives of this research are to:

1. Assess changes in the degree of forest fragmentation and connectivity in Virginia's landscapes between 2001 and 2011 and investigate effectiveness of satellite imagery at different spatial resolutions, for forest fragmentation and connectivity analysis.

2. Identify impacts of forest fragmentation and connectivity on the Wood Thrush and Ovenbird species in Virginia.
3. Assess the role of management policies on forest fragmentation and connectivity and biodiversity conservation.

The key outcome of this study provides a strategic decision tool to enable stakeholders and managers of Virginia's landscapes to plan development of the region while protecting its biodiversity. Identifying relevant drivers of forest fragmentation and connectivity, their effects, and the extent to which forest fragmentation and connectivity has occurred in Virginia, will provide managers and policy analysts with the scientific basis and tools necessary to promote landscape sustainability and biodiversity conservation particularly for the Wood Thrush and Ovenbird species.

1.2.2. Structure of Dissertation

There are three main objectives in this dissertation, revolving around landscape changes over time within Virginia. The broad aim and objectives of this research are introduced in *Chapter 1*. Data sources as well as key stakeholders, pertinent to this research are also identified in the first chapter.

In *Chapter 2*, I assess the degree of changes in forest fragmentation and connectivity in Virginia between 2001 and 2011 and investigate the effect of spatial resolution of different satellite imagery, for forest fragmentation and connectivity analysis. Results identify the trends in forest growth and preservation in Virginia and equip policy makers with important decision tools to conserve forests in Virginia.

Chapter 3 focuses on identifying the impact of forest fragmentation and connectivity on the Wood Thrush and Ovenbird species in Virginia. In this chapter, a correlation between population abundance of the Wood Thrush and Ovenbird species and the trends in forest fragmentation and connectivity in Virginia over a ten year period, is established.

Chapter 4 examines the role of management policies in forest fragmentation and connectivity within Virginia and how these policies can be used to conserve biodiversity. The aim of this chapter is to identify specific management policies that would be effective in conserving habitats with high ecological integrity and assess the role of riparian buffers as ecosystem corridors to promote habitat connectivity.

The overall conclusion of this research is presented in *Chapter 5*. This last chapter highlights the results from preceding chapters and the key information obtained. The stumbling blocks and bottle-necks in the approaches used in the research are identified and future avenues for continuing the research are presented.

1.3 Literature Review

1.3.1 Relationship between human activities, fragmentation and connectivity

As human populations across the world increase, numbers of people that live in urban areas also increases. Currently, the population growth rate in urban areas is approximately 2.4%, 1.7% higher than the population growth rate in rural areas (Wu and Murray, 2003). With over 45% of the world's population living in urban areas, there is a need to understand changes made to the natural landscape to convert natural lands into residential urban centers and the effect that this has on the sustainability of the natural environment. It has therefore become necessary to understand and manage the resulting natural and increasingly urban environment in order to ensure that biological species are not affected adversely.

Urbanization rates have been accelerating globally over the past several decades, as a result of increasing human populations, therefore becoming an increasingly important cause of forest habitat fragmentation (Liu et al., 2016). Between 2010 and 2050, it is estimated that urban population around the world will increase from the current 51.6% to 67.2% with the built environment increasing by three times (Liu et al., 2016). As urbanization increases, large areas of natural habitat, such as forests, are lost as they are converted into impervious surfaces, including residential areas, roads and railways.

Over the years, several studies have supported the fact that habitat fragmentation is strongly linked with anthropogenic activities. For instance, Fahrig (1997) showed a significant quadratic relationship between the number of patches formed in an area and habitat loss as a result of human activities. The results from both Gustafson and Parker (1992) and Pearson and Gardner (1997) showed strong relationships between human activities and rates of fragmentation. In Liu et al. (2016), fragmentation is shown to be strongly correlated with urbanization rates in a study conducted across 16 world cities.

According to a study by the Millennium Ecosystem Assessment (2005), human activities have fragmented more than half of temperate broadleaf and mixed forests. These changes to the natural landscape structure affect the movement of living organisms in these areas and ultimately affect the capacity of the natural habitats to provide key ecosystem services (Millennium Ecosystem Assessment, 2005). Relationships between connectivity of the landscape, which aids in the movement of organisms, and the structure of the habitats which includes the sizes and shapes of the patches, affect the usefulness of the ecosystem services concept for ecosystem and landscape management (Mitchell et al., 2013; Tscharntke et al., 2005 and Kremen et al., 2007). Figure 1.1 demonstrates the strong linkage between human activities, fragmentation and connectivity in natural landscapes.

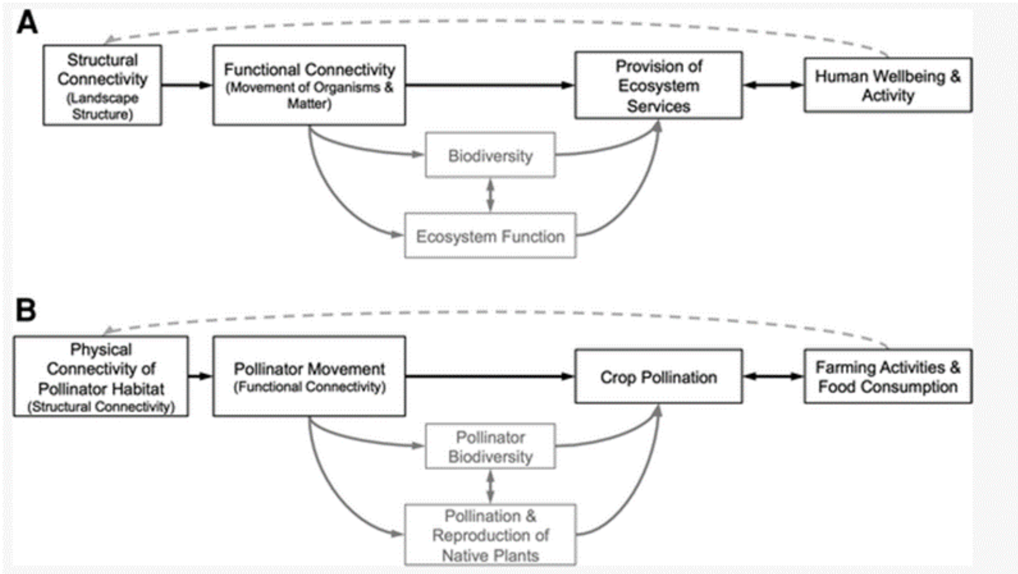

Figure 1.1. Relationship between human activities and forest fragmentation. Adapted from Mitchell et al. (2013). Generally, ecosystem function is influenced by human activities that impacts fragmentation and connectivity in natural landscapes as shown in (A). In terrestrial ecosystems such as forests (B), pollination services are shown to be influenced by landscape connectivity and fragmentation.

Landscape fragmentation and connectivity are correlated, with connectivity defining the degree to which fragmented patches within a landscape facilitate the movement of organisms and materials from one patch to another. Taylor et al. (1993) defined connectivity as the extent to which a given landscape facilitates movement of species and other ecological flows. Connectivity plays an instrumental part in biological diversity and conservation by counteracting effects of habitat fragmentation (Crooks and Sanjaya, 2006). A corridor is a link between two habitat patches that offers the possibility of dispersal between habitats and accommodates some species in the process (Saura and Rubio, 2010).

In any given landscape, connectivity determines those portions of the total habitat accessible for an organism located in a specific point in the landscape (Saura et al., 2011). Connectivity also takes into consideration the ability of species to move from one habitat area to another through non-habitat areas (Tischendorf and Fahrig, 2000). Therefore, connectivity is not only important within individual habitat units but also aids in movement of species between several habitat areas, serving as the link between different habitats (Manning et al., 2009; Saura et al., 2011).

1.3.2 Components of a Landscape

A landscape is composed of spatial elements, expressed popularly by Forman and Godron (1996) as the patch-corridor-matrix model. In this model, Forman and Godron (1996) identify three major landscape components (the ***patch***, the ***corridor***, and the ***matrix***) which they propose define the pattern of the landscape.

1.3.2.1 Patch

Wiens (1976) and Forman and Godron (1986) define a ***patch*** from an ecological perspective, as a discrete area or a period that is highly relevant to an individual organism and is characterized by similar environmental conditions that have distinct boundaries as a result of the discontinuity in its environmental character from its surroundings.

From *an organism-specific point of view*, a patch is an area within a landscape that has high connectivity for an organism of focus, meaning that the patch enables the organism to adequately interact with the landscape habitat structure (Girvetz and Greco, 2007; Taylor et al., 2006; Wiens, 2006)

In *spatial terms*, especially with GIS, a patch is a cluster of similar pixels that are grouped together in a substantial size that can support the survival of biological species (Beier et al., 2013). The term 'substantial' is used loosely to depict an area that is big enough to at least support one breeding unit (Beier et al., 2013).

These definitions of habitat patches suggest that specific organisms in the landscape determine the structure of patches in the landscape. Thus, there are diverse patches in a landscape, dependent on the perceptual abilities as well as behavioral responses of the organisms in that landscape (Girvetz and Greco, 2007). Patches are therefore only useful spatial concepts relative to individual organisms and their environmental ecology, rather than fixed elements of the landscape (Turner et al. 2001).

1.3.2.2 Corridor

Another key element of the landscape is ***corridors***. Corridors are defined as narrow strips of land that are unique in that they are different from the surrounding matrix (Forman and Godron, 1986). Corridors, though often isolated, are usually attached to patches with similar characteristics, serving as links between patches (McGarigal, 2015). Functionally, corridors are very diverse, serving variously as dispersal conduits, habitats or barriers (McGarigal, 2015).

Whereas habitat corridors provide temporary or permanent survivorship and movement for certain species, barrier habitats impede the movement of species and the flow of energy across a landscape (McGarigal, 2015). McGarigal (2015) highlighted the fact that corridors are not distinguishable based on their structure alone, but most importantly, by their function which is dependent on the organisms present in the landscape.

Corridors frequently are defined inappropriately. According to Beier and Noss (1998), a corridor is a linear habitat embedded in a dissimilar matrix and serves as the connection between two or more larger blocks of habitat, aiding in conservation because of their capacity to maintain and enhance the viability of specific species' populations in those habitats.

The importance of corridors stems from the habitats that they are connecting. The full potential of corridors are realized when there are discontinuous natural areas. Human activities often aid in corridor creation and are most likely the reasons why corridors are so important in landscapes. Anthropogenic activities create habitats with unsafe migration or dispersal channels, resulting in lower biodiversity in these habitats. Corridors then become necessary for the continuous survival of the species within the landscape.

1.3.2.3 Matrix

A ***matrix*** plays the dominant role in determining how a landscape functions. According to Forman and Godron (1986), a matrix is the most extensive and most connected landscape element type. A matrix is defined as a spatial assemblage of habitat patches and the associated land uses that form a background system (Forman and Godron, 1986).

In any given landscape, the matrix is the greatest in areal extent and is most often very obvious to the researcher. In other cases, however, the matrix is not very obvious as a landscape may contain two or more dominant elements with similar areal extents. The scale of investigation is also a key factor in identifying the matrix in a landscape.

The matrix in a given landscape is an important element in understanding the landscape pattern. In effectively managing any landscape, identifying the matrix is an important step that if not paid attention to, could result in the failure of the landscape to support biodiversity. Matrix determines the level of heterogeneity in a landscape as it connects habitats that are of different shapes and sizes (Barnes, 1999). In any given ecological system, a matrix has the highest degree of connectivity. Thus, for a forest matrix with fewer gaps or patches, there will be higher connectivity compared to a forest matrix with more patches.

The structure of a matrix is characterized by the density of its patches, boundary shape, networks, and heterogeneity. Patch density within a matrix is important for animal movement, while the shape of the boundaries have significant implications for neotropical migrant birds (Barnes, 2000). Although the landscape matrix is made up of patches, matrixes differ from patches in structure and function.

1.3.3 Causes of forest habitat fragmentation

The process of converting long continuous patches of habitats into smaller heterogeneous patches, referred to as habitat fragmentation, is caused by many factors. This dissertation will focus on anthropogenic causes of fragmentation.

1.3.3.1 Agriculture

In Virginia, agriculture plays a significant role in the wealth of the community. Through agriculture, there is a market base created to buy and sell local produce and also diversify the products and services for commuters and visitors to the community (Scott and Walker, 2016). In Virginia, agriculture is the oldest and the principal industry, producing approximately $52 billion every year in cash (Scott and Walker, 2016). Giles, Montgomery, and Pulaski counties, 3 out of the 95 counties located in Virginia, have a total of 1,426 farms that cover nearly 270,000 acres of land (Scott and Walker, 2016). Agricultural produce in Virginia is very diverse, varying from corn, sweet potatoes, a significant amount of dairy, meat, poultry, soybean, wheat, tomatoes, apples, barley to rye among others.

From 2010, there has been a 34% increase in the quantity of exported goods and services in the United States with agricultural exports, increasing up to about 40% within the same time period (Farm Bureau, 2016). Despite the importance of agriculture to development, uncontrolled expansion of agriculture leads to landscape fragmentation and degradation that can result in the reduction of biodiversity. It is important to strive for sustainable development by controlling the expansion of all land uses in the New River Valley.

1.3.3.2 Deforestation

This process involves the change or conversion of land areas predominantly covered by forest trees to non-forest land use such as agricultural land, residential areas or barren and transitional lands (Tejaswi, 2007). The FAO has determined a specific threshold of 10% to identify if a specific area can be termed deforested or not; if the tree canopy of a once forested land area is converted to another land use below the 10% threshold, the area has definitely undergone deforestation (FAO, 2000).

Factors that result in deforestation can either be intentional or accidental (Tejaswi, 2007). For instance, uncontrolled grazing that leads to the inability of young trees in the areas to naturally regenerate is an unintentional cause of deforestation. Another case of unintentional cause of deforestation is through uncontrolled fire disturbances. However, the most common causes of deforestation are purposely conducted by man through activities such as timber logging and residential development (Tejaswi, 2007).

Figure 2 below shows that the causes of deforestation are varied stemming from a range of socio-economic factors and differ across continents. Breakdown in cultural systems might result in the inefficiency of upheld social institutions that result in land ownership conflicts, which lead to deforestation (Islam and Sato, 2012). Also, weak governmental legislation can be a significant factor in the rate of deforestation.

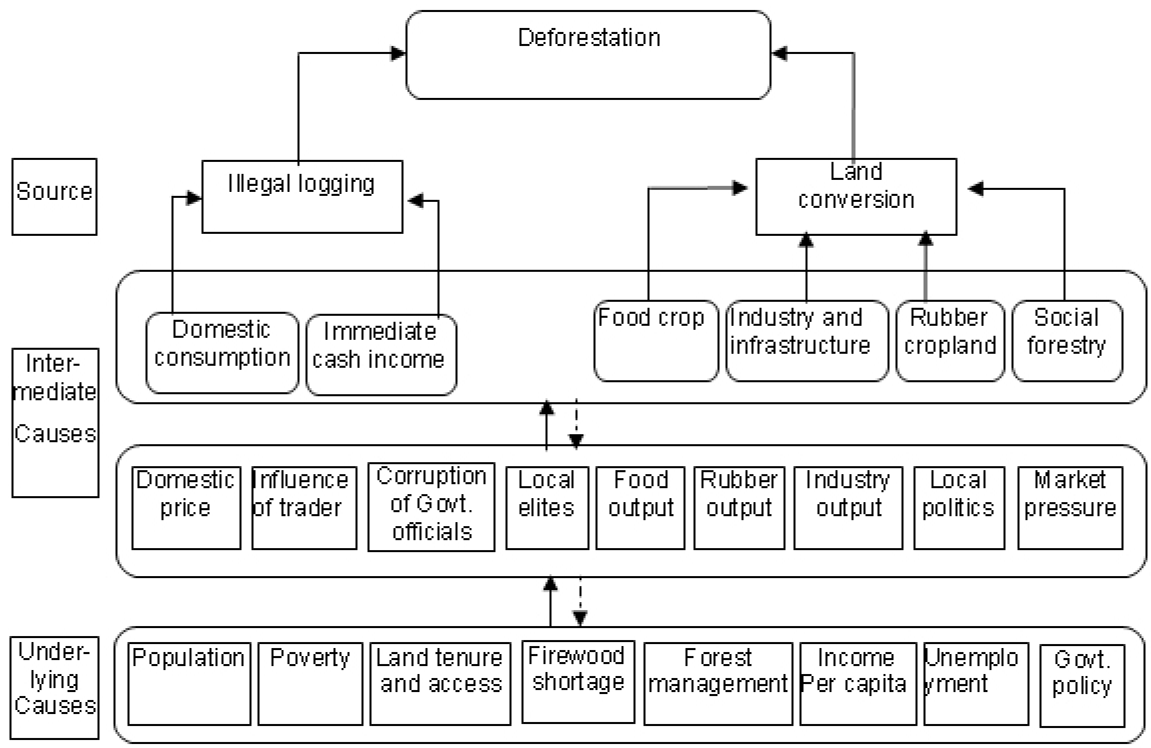

Figure 1.2. Adapted from Islam and Sato. (2012). Different factors affect deforestation. The underlying causes are important for the identification of solutions to the more obvious problems.

As the rate of deforestation increases, the rate of forest fragmentation, the process resulting in the conversion of previously continuous forest lands into patches of forest disconnected by non-forested lands, consequently increases (Tejaswi, 2007). Large patches of forest lands are replaced by smaller patches separated by diverse land cover types such as agricultural lands and residential lands; this conversion significantly impacts the inhabitants of forest land cover types.

1.3.3.3 Mining

Frelich (2013) identifies mining as one of the major environmental impacts on the global boreal forest, as a result of widespread presence of ancient rock formations containing metallic ores. The impacts of mining on forest habitats cannot be oversimplified as the ecological footprint of mining activity spreads beyond the area directly impacted (Frelich, 2014). The primary footprint of mining includes the area directly impacted by the mine, processing facilities, roads and the energy transmission network built to facilitate the mining activities (Frelich, 2014). The secondary footprint comprises the surrounding areas impacted indirectly by the mining activities such as the areas that receive acidic rain, that cause changes in the landscape across diverse distances including forest fragmentation (Frelich, 2014).

Mining can directly displace forests and possibly change the composition of living organisms in any forest remnants with the magnitude of its impacts dependent on the forest acreage available for commercial timber provision (Frelich, 2014). The type of forest can also change as a result of mining since the division of large forest areas into smaller patches favors early successional and generalist species such as poplars and red maple trees.

1.3.3.4 Pollution

Air pollution can be a significant cause of forest habitat fragmentation as pollution of the air in areas near harmful emissions can lead to atmospheric acidification. As a result, acid rain in particular sections of the forest can damage forest vegetation cover, leading to forest fragmentation. It is possible for acidic compounds present in acid rain to extensively damage vital forest ecosystems and biodiversity. Deposition of acid ions lead to loss of tree leaves that create forest canopy gaps and could lead to the complete die-off of specific forest tree species (Hames et al., 2002).

Land pollution, as a result of the discharge of various kinds of chemicals on areas close to forest regions, can make the environment incapable of supporting certain specialist forest species including both trees and animals (Hames et al., 2002). The result is fragmentation of large forested areas into smaller patches (Hames et al., 2002). This interferes with the interactive food chains of the living organisms in the forests since the chemicals contaminate the food of some of the living organisms in the area.

1.3.4 Effects of forest fragmentation

McGarigal et al. (2002) lists at least 40 measures of forest fragmentation with some of them interacting to result in similar effects. The process of habitat fragmentation results in four main effects: 1) increase in number of patches and patch shape complexity, 2) decrease in patch sizes, 3) increased edge effects and 4) increase in isolation of patches.

1.3.4.1 Reduction in patch area

As forests become fragmented, over time, many more small patches are formed compared to larger patches (Ranta et al., 1998; Ewers and Didham, 2007). Large patches often correspond to larger habitat area (Fernandez-Juricic, 2000) and therefore, generally, smaller patches contain fewer species than do larger patches (Debinski & Holt, 2000). As patches decline in size, resources available for biological species in that area, become limited, leading to a decline in colonization rates of newer species and lower reproductive rates among already present species. For example, Donovan et al. (1995) found that forest birds had lower reproductive rates in small patches than in large patches. As such, there is an increase in the extinction rate of species in small areas (Hanski and Ovaskein, 2000).

1.3.4.2 Change in patch shapes and increase in number of patches

Shape complexity occurs as a result of forest fragmentation over a long period of time. Laurance and Yensen (1991) explains that forest patches with high complexity have higher degrees of fragmentation, compared to those with simpler shapes. The effect of shape complexity, as a result of forest fragmentation, on biodiversity varies. As patches become increasingly complex, colonization and emigration rates within the patches change as a result of mobility restrictions, causing variability in species population size (Cumming, 2002). For patch specialists such as those species that prefer core areas, increasing shape complexity is a problem because core habitat areas decrease with increasing patch complexity (Didham, 2010). The effect of patch complexity on patch corridors that are linearly structured, such as roads, is likely to be much more severe (Didham, 2010).

A common effect of forest fragmentation is an increase in the number of forest patches within an area (Fahrig, 2003). As large continuous forest areas divide into many smaller patches, at some point each single patch will be too small to sustain a local population (Fahrig, 2003). Ultimately, the population size of local species will be reduced, decreasing the probability of persistence of that species.

1.3.4.3 Increased edge effects

Forest fragmentation leads to increased edge area, meaning that more of core patch areas, are exposed to external influences (Didham, 2010). Cadenasso et al. (2003) described edge effects as the transition in abiotic and biotic variables that occurs across the boundary between adjacent land-use types. Edge areas have different conditions compared with core areas; edges are typically drier, windier, and hotter and have brighter light intensity (Harper et al., 2005; Didham, 2010). Such changes determine types of species that can exist in those areas. Hence, edge effects could have both negative and positive effects on species population size (Fahrig, 2003). Habitat specialists such as ovenbirds that prefer core forest areas decline in population size as edge areas increase due to forest fragmentation. Species that prefer to live in edge areas, on the other hand, increase in population size. Another direct effect of increased edge effects is increased predation. Chalfoun et al. (2002), Kuitenen and Helle (1988) and Andrén (1995) all demonstrated that forest birds experienced increased predation with increasing edge areas.

1.3.4.4 Increased isolation

Isolation is a measure of the lack of habitat in the landscape surrounding the patch of interest (Fahrig, 2003). As forest fragmentation increases, patch isolation also increases, leading to the inability of species to maintain viable populations. Thus, species richness will decrease with time through a process called ‘species relaxation’ (Brooks et al., 1999). Patch isolation is also strongly correlated with species movement. Bender et al. (2003) and Tischendorf et al. (2003) showed that, with increasing patch isolation, movement of species between patches decrease, leading to inbreeding, the phenomenon in which closely related individuals within a population mate. Inbreeding as a result of patch isolation leads to a reduction in overall fitness of a local population because of increased probability of offspring being affected by deleterious traits (Bender et al., 2003).

1.4 Description of Data

Data required for this research, as well as their source, are listed in Table 1 below.

Table 1.1. Source of data for dissertation

DATA	**SOURCE**
Landsat imagery of Virginia	USGS
Sentinel imagery of Virginia	USGS
UAV imagery of Virginia	Virginia Tech
Aerial imagery of Virginia	USGS, Virginia Geographic Information Network
Data on bird counts	Audubon Society
Hydrological data of Virginia	USGS
Population data for Virginia	United States Census Bureau
Virginia Natural Landscape Assessment (VaNLA) product	Virginia Department of Conservation and Recreation
Data on land conservation in Virginia	Virginia Department of Conservation and Recreation

1.5 Study Site

The state of Virginia forms a part of both the Southern and the Mid-Atlantic United States. Most of Virginia's rivers including the Potomac, Rappahannock, York, and James, flow into the Chesapeake Bay, which separates the mainland part of the state from the Eastern Shore, consisting of Accomack and

Northampton counties. West of the eastern continental divide, the New River forms the principal drainage.

Virginia presents an ideal case study for anthropogenic environmental disturbances as a result of its high population growth rate, currently estimated at about 5% annually (Decennial Census and 2016 State Population Estimates). The remarkably high rate at which the state of Virginia is changing, from forest lands to agricultural lands, and then, more recently, to residential lands, calls for examination of effects that these changes might have on biological diversity.

Forest cover of the state of Virginia is currently estimated at about 15 million acres, making up two-thirds of the total Virginia land cover (Barrett et al., 2012). In managing these forests, logging operations form a critical component, as woodlots provide important income to forest landowners (Bolding et al., 2010). Rephann (2008) noted that the forest cover in Virginia provides an estimated annual economic output of $23 billion while creating about 145,000 forest-related jobs.

The diversity of Virginia is reflected in its varied physiographic regions and forest types. Fleming et al. (2013) noted that Virginia has one of the most diverse landscapes in the eastern United States, varying from its sandy, low-elevation soils of the Coastal Plain to dry, rocky soils on the southwestern slopes of the Appalachian Mountains. Such diversity leads to a wide range of forest types in the state with most heavily influenced by anthropogenic disturbances such as fire, logging, and farming. Only 2% of forest land in Virginia can be considered old growth, characterized by minimal human disturbance, diverse forest canopy structure, and the presence of old trees, coarse woody debris, and dead standing trees (Gagnon, 2016).

Virginia's forests include Oak-Pine, Pine, Bottomland Mixed Hardwood and Oak-Hickory, which form the most common forests in the state, accounting for over 61% of the total forest cover (Gagnon, 2016; Rose, 2015). During the 17th and 18th centuries, when Europeans first arrived in Virginia, a history of timbering, deforestation and agriculture began in the state that changed the land cover structure as well as watershed of Virginia (Themes, 2016). Before this event, over 90% of Virginia was covered with trees including chestnut, black walnut, hickory, a variety of oaks, and pines. However, large farm operations began, leading to the planting of tobacco and the felling of hardwood trees, valued for their economic return in Europe (Themes, 2016). The felling of trees for timber led to high deforestation rates and forest fragmentation in Virginia.

Following the era of wide scale agricultural expansion and consequent high deforestation rates in Virginia, *reforestation* trends, as a means of meeting high timber demands, began (Gao and Yu, 2014). Agriculture was abandoned and people migrated from the rural area to the urban centers as a result of this economic shift (Grimm et al., 2008). Urban expansion led to rapid suburban development which drove environmental changes at local, regional, and global scales and impacted biodiversity. Chambers et al. (2007) found that the extension of urban sprawl to distant suburbs caused landscape fragmentation. Fragmentation in turn, enhances edge effects that alter mass, energy and information flows within land types and therefore impact negatively on the biodiversity present (Gao and Yang, 1997; Gao et al., 2004). As a result of high urbanization rates in Virginia which may prompt fragmentation, it is important to assess the extent of forest fragmentation and its impacts on biodiversity, in Virginia.

Forest fragmentation is one of the biggest threats to the health of Virginia's forests (Gagnon, 2016). Forest fragmentation in Virginia affects the function of forest systems at the landscape level and reduces land-use options for the future (Gagnon, 2016). The Virginia Department of Forestry (VDOF 2014) showed that, since 1997, more than half a million acres of forestland in Virginia have been converted to other uses. The study predicts that Virginia will lose 1 million acres of forest in the next 25 years if current development trends continue. Such changes have significant impacts on Virginia's biodiversity.

Fragmentary loss of forestlands in Virginia is not the only concern of ecologists. Air pollution in Virginia also raises questions about the sustainability of the forest resources. Currently, the Radford Army Ammunition plant, which supplies energetics, propellants and munitions nationally, is the second largest employer within Pulaski County. The Environmental Protection Agency has ranked the plant as the leading cause of air and water pollution in the commonwealth over the last thirteen years (EPA, 2014). The increase in pollution in the region will undoubtedly lead to the need for species to migrate, making the establishment of corridors very significant in the survival of terrestrial species, including the Wood thrush and Ovenbird (Roth et al., 1996; Morimoto at al., 2012).

The state of Virginia has a rich mining history, spanning decades and contributing to the early development of the region. For instance, the Chesapeake and Ohio railroad was constructed in 1873 to provide a faster travel route for miners that commuted between the Atlantic Coast and the Ohio River Valley. With the construction of this railroad, human population in the region at the time doubled. More mining sites were consequently created to aid in coal production. Creation of these mining sites resulted in the destruction of many natural habitats, ultimately contributing to landscape fragmentation. Hence, this region forms a fruitful site for study of habitat fragmentation, connectivity and its biological impacts.

1.6 Expected Research Impact

Forest habitat fragmentation leads to alteration of ecosystem functioning (Hill et al., 2011). For example, feeding guild compositions of avian populations in an area change following habitat fragmentation (Hill et al., 2011). Small forest fragments are more likely to experience increased habitat disturbance and altered micro-climates (Benedick et al., 2007) as well as changes in trophic organization which may affect the long-term viability of avian species. This study is therefore important to provide policy makers with a strategic decision tool by informing them of the extent of forest fragmentation in Virginia and consequences of this pattern on Wood Thrush and Ovenbird species. This way, development decisions are made with conservation in mind.

Looking at fragmentation from a state-wide perspective is important because forestry laws and policies, although varying widely from state to state, are uniform within a given state. Thus, use of Landsat imagery spanning wide areas and bird data, which is state-specific, makes it easier for policy makers who consider the entire state, and not specific areas of the state, to make choices that are more informed. On the other hand, forest fragmentation data provide contextual information (Lister et al., 2003). This means that forests in different parts of the state will have different characteristics. Thus, identifying and interpreting land cover uses within the context of landscape configuration, made possible via the use of moderate to fine resolution imagery from Virginia base map and UAS, will provide better understanding of the status of forest resources and more effectively, aid regional planners and decision makers.

Despite knowledge of the significant differences in fragmentation metrics from images of different spatial resolutions, effects of spatial resolution on fragmentation metrics is not fully understood. Forest fragmentation studies such as McGarigal and Marks (1995) have cautioned against comparison of fragmentation metrics obtained from images with varying spatial resolutions. This lack of comparability limits the importance of quantitative forest fragmentation analysis (Saura, 2004). This study therefore, provides further insight to effects of spatial resolution on forest fragmentation metrics and identifying those metrics that can be compared across images of differing spatial resolutions.

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Chapter 2: DEGREE OF FOREST FRAGMENTATION AND CONNECTIVITY IN VIRGINIA BETWEEN 2001 AND 2011

2.1 Introduction

This chapter describes effects of spatial resolution of remote sensing images on the calculation of landscape metrics commonly used in forest fragmentation studies. Geospatial information on the size, shapes and areas of forested land areas in Virginia over a ten-year period, 2001 to 2011, is analyzed to determine the degree of change in forest fragmentation as well as corridors in the region. Effects of spatial resolution of images used in fragmentation analyses are also reviewed in this chapter.

2.2. Publications

Two manuscripts related to this chapter were published in the Journal of Landscape Ecology and can be found in the Appendix A and B.

1. Fynn, I. E. M. and Campbell, J. 2018. Forest fragmentation and connectivity in Virginia between 2001 and 2011. Journal of Landscape Ecology. 11(3): 98 – 119.
2. Fynn, I. E. M. and Campbell, J. 2019. Forest fragmentation analysis from multiple imaging formats. Journal of Landscape Ecology. 12(1): 1- 15.

Chapter 3: IMPACT OF FOREST FRAGMENTATION AND CONNECTIVITY ON WOOD THRUSH AND OVENBIRD

3.1 Introduction

This chapter identifies the significance of possible variables that determine the population sizes of Wood Thrush and Ovenbirds in Virginia. Variables such as topographic characteristics and changes in forest fragmentation in Virginia are analyzed using the MCMM package in R software to determine how they influence bird populations.

3.2. Manuscript

Appendix C is the draft of the original manuscript being prepared for publication in *Condor* in May.

Appendix D contains the appendixes for Appendix C.

Chapter 4: THE IMPACT OF MANAGEMENT POLICIES ON FRAGMENTATION AND CONNECTIVITY

4.1 Introduction

This chapter identifies management policies that will be effective in conserving biodiversity in Virginia. The effect of population density on ecological integrity of patches is examined by comparing the patch metrics of densely populated counties with those of sparsely populated counties. Analysis on the number of ecological patches that are not managed by either the federal, state, local or private bodies was also carried out. The capacity of riparian buffers as effective corridors is examined and their feasibility explored by analyzing the responses of landowners and farmers in Virginia.

4.2 Manuscript

Appendix E is the draft of the original manuscript being prepared for submission to the *Journal of Environmental Quality* in July.

Appendix F is the survey questionnaire sent to farmers and landowners in Virginia

Chapter 5: OVERALL CONCLUSIONS

Human activities have both positive and negative effects on natural ecosystems. While anthropogenic activities such as timber harvest and road construction cause forest fragmentation, other human efforts such as intentional establishment of riparian buffers that have the capacity to serve as corridors can improve connectivity between natural ecosystems such as forests. Scientific analysis involving remote sensing and geospatial technologies help to identify changes in natural ecosystems, effective approaches to understand human influence on natural ecosystems, and ways to improve sustainability of natural ecosystems.

Using Virginia as a study area, this dissertation has focused on:

1) Assessing the degree of forest fragmentation and connectivity in Virginia's landscape between 2001 and 2011 and investigating the effectiveness of satellite imagery at different spatial resolution for forest fragmentation and connectivity analysis,

2) Identifying the impact of forest fragmentation and connectivity on Wood Thrush and Ovenbird species in Virginia, and

3) Assessing the role of management policies in forest fragmentation and connectivity and in biodiversity conservation.

While forest fragmentation is generally accepted to have negative consequences on biodiversity conservation, its effect on specific species in Virginia had not yet been established. This dissertation uses a Markov Chain Monte Carlo statistical approach to establish effects of forest fragmentation, connectivity, and environmental variables, such as slope and characteristics of the physiographic regions in Virginia, on the populations of Wood Thrush and Ovenbirds. Results of this chapter show that Ovenbird populations are negatively affected at local scales by fragmentation and slope and are more abundant in the Coastal Plain region. At the regional scale, none of the variables are considered to significantly affect Ovenbird populations. The model shows that Wood Thrush populations declined significantly in 2011. The methodology used in this study can be replicated in other areas with different species.

Results of this dissertation are useful for understanding forest fragmentation and connectivity, as well as their impact on biodiversity. The first part of the dissertation assessed the extent to which forests in Virginia have either become fragmented or connected over a ten-year time period between 2001 and 2011. The results established the fact that forests in Virginia increasingly are becoming fragmented. This dissertation also showed that results of forest fragmentation and connectivity analyses are dependent on the spatial resolutions of the satellite imagery used, and that researchers should be cautious in justifying use of specific satellite imagery for differing purposes including forest classifications.

Despite the fact that spatial resolution of remote sensing images may impact forest fragmentation results as corroborated by other studies, it does not appear that spatial resolution alone may suffice to explain the large differences in the results of this study. These large differences may be due partly to incorrect classification of the forest classes. To improve future results, considerable thought should be devoted to the classification method used. Differing classification methods should be assessed to identify the most accurate methods to provide a strong justification for the results.

Not only did this dissertation identify effects of forest fragmentation and connectivity, it also proposed effective management policies that could be useful in conserving biodiversity. Specifically, this study showed differences in characteristics of ecological zones in densely and sparsely populated counties in Virginia and that management efforts need improvement as many ecological zones in Virginia are currently unmanaged, and are therefore at risk of destruction. This study illustrated how to create corridors that connect ecological patches, which could be useful in promoting interactions among wildlife populations and therefore ensure persistence of healthy species populations. To understand how a riparian buffer policy can promote conservation of biodiversity, this study reported opinions of a sample of Virginia landowners and farmers. This dissertation showed the importance of financial incentives and education in the adoption of best management policies such as a riparian buffer policy. While this study provides an essential framework for understanding forest fragmentation and connectivity as well as their effects on biodiversity, it also encourages discussion of the importance of selecting the appropriate satellite imagery for specific research tasks especially those involving forest classifications. This research also highlights gaps in management of sensitive ecological areas in Virginia, and the need for further research to improve understanding of land ownership, the use of natural lands and its effects on biodiversity conservation.

Appendices

Appendix A

Fynn, I. E. M. and Campbell, J. 2019. Forest fragmentation and connectivity in Virginia between 2001 and 2011. Journal of Landscape Ecology. 11(3): 98 – 119.

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FOREST FRAGMENTATION AND CONNECTIVITY IN VIRGINIA BETWEEN 2001 AND 2011

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ABSTRACT

With an annual population growth rate currently estimated at about 5 %, Virginia presents an ideal case study for anthropogenic environmental disturbances. Urbanization as a result of increasing human activities has led to fragmentation of many crucial habitats, especially forests. Analysis of the extent to which forest fragmentation and connectivity have occurred in Virginia and corresponding changes associated with these processes, is relevant for conserving forest habitats and the biodiversity that they support. This study applies FRAGSTATS, a software system developed to assess forest fragmentation and connectivity, in combination with ArcGIS, to identify changes in forest patch metrics for Virginia over a ten-year interval (2001, 2006 and 2011) using National Land Cover Datasets (NLCD) maps as data source. Results show that, over ten years, forest patches have significantly declined in size, while the number of forest patches and total length of edge areas have increased over time. Results of this study show that road density in Virginia has no significant effect on forest fragmentation between 2001 and 2011. Analysis using ArcGIS revealed that sizes of core forest areas in Virginia are declining, and that these reductions match local topographic slope. This is because the steepness of the slope of an area dictates the degree of human activities in that area. These results suggest that urban sprawl associated with areas with gentler slopes, may have significant, long-term consequences for natural forest ecosystems and ultimately, biodiversity conservation.

Keywords: Forest Fragmentation, FRAGSTATS, Core, Connected, Fragmented, Forest patches, Patch metrics, Edge.

INTRODUCTION

Natural landscapes are essential for maintaining habitat quality and ecosystem services. Recent research shows that natural landscapes, such as intact forests, are significant for mitigating climate change, maintaining water supplies, conserving biodiversity, and protecting human health (Watson *et al.*, 2018). It is therefore important to know the degree to which expansion of the terrestrial human footprint affects natural forests and corresponding changes in environmental values.

Forest ecosystems contribute significantly to biodiversity conservation and over time, have evolved their resilience and adaptation capabilities to both anthropogenic and natural disturbances such as drought, fire and residential construction (Noce *et al.*, 2017).

Conservation of forest ecosystems will be more effective if impacts of changes to forest distribution due to environmental disturbances, are taken into consideration (Leroux & Rayfield, 2014; Socha *et al*., 2016). Thus, in planning of conservation strategies, changes in species habitats and the factors affecting the composition of forest ecosystems over time, are critical (Serra-Diaz *et al*., 2015). Studies have shown that changes to forest ecosystems may occur over either a short period of time or may span hundreds of years, accompanied by species migration and evolutionary adaptations (Noce *et al*., 2017; Trumbore *et al*., 2015). Current research has also demonstrated that the changes in forest ecosystems are significantly correlated with human activities, climate and environmental conditions and topographic factors (Williams *et al.,* 2012; Bellingham & Tanner, 2000; Noce *et al.,* 2017).

Over the last hundred years, research conducted by the U.S. Department of Agriculture (USDA) Forest Service shows that the amount of forest land in the United States, has remained nearly constant (USDA Forest Service, 2012). A worrying trend, however, is the change in distributions of forests between different regions across the country as a direct result of human activities (USDA Forest Service, 2012). Forest fragmentation describes the process by which forested areas are reduced to smaller patches interspersed with non-forest land cover. Forest fragmentation in a particular area across a time gradient is affected by patterns of forest gain and forest loss, which invariably means that forest fragmentation can occur even if the absolute area of the forest land in a given region does not change.

Forest fragmentation forms a significant aspect of the distribution of forest systems as forest species, especially specialists, are adapted to forest habitat conditions, most particularly to either edge or interior forest habitats. As a result, changes in the patterns and extent of forest fragmentation, affect population sizes of forest species such as birds and other mammals (Fahrig, 2003). For example, increasing forest fragmentation leads to more pronounced edge effects, and to decreasing numbers of interior-adapted forest species such as the ovenbird (*Seiurus aurocapilla*) and wood thrush (*Hylocichla mustelina*) (Donovan & Flather, 2002).

Changes in landscape characteristics, and for that matter, changes in forest fragmentation, occur over an indefinite amount of time, with some changes occurring abruptly and the effects of other changes becoming more visible after several years (Bennett & Sanders, 2010). Landscape changes as a result of forest fragmentation include: (a) declining areas of total forest land in a given region, (b) reductions in sizes of forest patches, (c) increased isolation of forest patches, and (d) changes in shapes of resulting forest patches from more curvilinear boundaries to straight edges (Bennett & Sanders, 2010). Increases in lengths of edge areas are another common consequence of forest fragmentation with detrimental influences on population sizes of specialist species, as a result of increased residential and other anthropogenic developments.

Rapid residential and urban development in Virginia raises concerns about effects of human activity on environmental quality, forest fragmentation and loss of biological diversity. For example, Virginia's Loudoun County, the third most rapidly growing county in the United States, experienced a population growth rate of 55 % between 1990 and 1997 (GWU, 1999). Formerly primarily agrarian, forest lands have gradually been altered in the county, making way for extensive impervious construction (Fuller, 2001). Such effects are not limited to Loudoun County, as similar trends have been identified across the state of Virginia. These effects have been of particular concern to conservationists as division of previously contiguous natural landscapes into smaller, more fragmented, patches of forest, lead to changes in climatic conditions, hydrology, soil and topographic characteristics of the area, which consequently have significant impact on biodiversity.

These concerns have increased interests in monitoring of environmental indicators such as forest cover changes, so that decision-makers in the state of Virginia can take action to mitigate impacts of urban development. Ricketts *et al*. (1999) suggest that areas with higher levels of human threat need to be identified and conservation efforts concentrated in those areas. Understanding relationships between forest fragmentation, human activities, and species population dynamics, is important for biodiversity conservation (Boulinier *at el*., 1998; Martin, 1998; Hanski, 1999). This study therefore aims to 1) assess the extent to which anthropogenic activities impact the state of Virginia and 2) quantify changes that have occurred in Virginia due to forest fragmentation between 2001 and 2011.

To quantify spatial patterns, measure the extent of forest fragmentation, and assess changes in a given area, McGarigal & Marks (1994) have developed a variety of indices that take into account the above listed changes. These indices measure the extent of forest fragmentation by tracking numbers of forest patches over a time gradient, aggregating resulting patches across that time gradient and assessing the complexity of the shapes of the resulting patches (Bennett & Sanders, 2010). These indices also measure connectivity between forest patches in a region, taking into consideration distances between the forest patches.

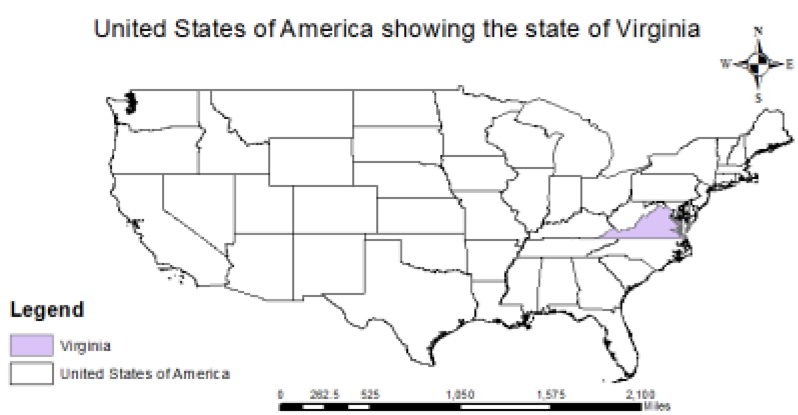
In this paper, we consider the extent of forest fragmentation in Virginia over a ten-year period between 2001 and 2011. For an in-depth analysis, we also examine changes in specific forest types in Virginia to identify which types of forests are undergoing the most changes over the study period. We also look at changes in sizes and shapes of forest habitat patches across the state, the degree of change in aggregation of patches over the ten-year period, as well as factors that might be contributing to forest fragmentation across Virginia.

METHODS

Study area

The state of Virginia forms a part of both the Southern and Mid-Atlantic United States (Fig. 1). Many of Virginia's rivers, including the Potomac, Rappahannock, York, and James, flow into the Chesapeake Bay, which separates the mainland from the Eastern Shore of Virginia, consisting of Accomack and Northampton counties.

Fig. 1: Map of the United States of America with Virginia highlighted. Virginia is one of 50 states in America.



Forest cover of the state of Virginia is currently estimated at about 15 million acres (approximately 6 million hectares), making up two-thirds of Virginia's total land cover (Barrett *et al*., 2012). In managing these forests, logging operations form a critical component, as timber provides important income to forest landowners (Bolding *et al*., 2010). Virginia presents an ideal case study for forest fragmentation, as a result of its high population growth rate, currently estimated at about 5 % annually (Decennial Census and 2016 State Population Estimates). The remarkably high rate at which the state of Virginia is changing, from forest lands to agricultural lands, and then, more recently, to residential lands, calls for examination of impacts that these changes might have on biological diversity.

Following the era of broad-scale agricultural expansion in the early 1900s and consequent high deforestation rates in Virginia, reforestation trends, as a means of meeting high timber demands, began (Gao & Yu, 2014). Agriculture was abandoned and people migrated from rural areas to urban centers as a result of this economic shift (Grimm *et al*., 2008). Urban expansion led to rapid suburban development which drove environmental changes at local, regional, and global scales and impacted biodiversity. Chambers *et al*. (2007) found that the extension of urban sprawl to distant suburbs in tropical forests caused landscape fragmentation due to altered precipitation regimes and land-use patterns. In the United States, Xie *et al*. (2015) were able to show how the forest structure in New England is significantly influenced by rainfall patterns and temperature. As a result of high urbanization rates in Virginia which may prompt fragmentation, it is important to assess the extent of forest fragmentation and its impacts on biodiversity, in Virginia.

Data

2001, 2006, and 2011 National Land Cover Database (NLCD), constructed from 30×30 m resolution satellite imagery (Jin *et al*., 2013), were used. The 2001, 2006 and 2011 NLCD maps were used because they were created using the same classification techniques (unlike the 1992 NLCD map), and are therefore comparable (Graham & Congalton, 2009). The NLCD has four main forest cover classes (deciduous, evergreen, mixed, and woody wetland forest), which were analyzed individually and also, combined into a single forest class for further analysis. The remaining NLCD classes were also combined into a single non-forest class. The raster forms of the NLCD dataset were used since raster data is the only form of data that can be analyzed in FRAGSTATS.

A Digital Elevation Model (DEM) of the state of Virginia with a 30 m resolution, obtained from the US Geological Survey, was used to derive elevation and slope maps of the state. Data on primary and secondary roads in Virginia, last modified in 2015, was obtained from the United States Development Agency (USDA). This information was used to create a road density map for the state to identify the impacts of roads on forest fragmentation and connectivity.

Methodology

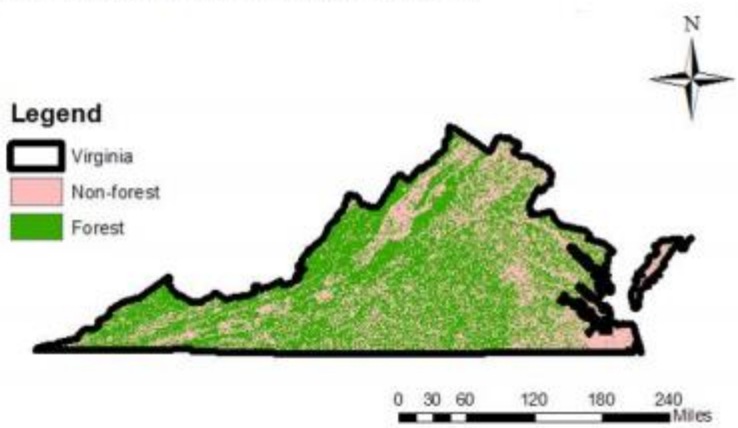
FRAGSTATS, a software system that provides detailed statistical information was chosen as one of the programs used to quantify forest fragmentation in this analysis (McGarigal & Marks, 1995). FRAGSTATS has the capacity to provide metrics such as area, density, total edge, and radius of gyration, at the individual patch level (Table 1). Similar metrics could be computed at the landscape and also, class levels. While landscape level metrics involves different land cover types such as forest, urban land and water, class level metrics emphasize specific land cover types, aggregating the patches as a single unit within the landscape instead of individual units that patch metrics is useful for. However, for this analysis, patch metrics within the forest class in Virginia, were the most important metric to determine how forest patches were changing over the ten-year time period.

Table 1: Patch metrics calculated in FRAGSTATS (McGarigal & Marks, 1995).
These patch metrics are important indicators of forest fragmentation and connectivity.

Patch Metric	Description
Patch Density	The number of patches divided by total landscape area.
Largest Patch Index	The area of the largest patch divided by the total landscape area, multiplied by 100 (to convert to a percentage)
Edge Density	The sum of the lengths (m) of all edge segments, divided by the total landscape area (m^2)
Landscape Shape Index	The sum of the landscape boundary and all edge segments (m) within the landscape boundary, divided by the square root of the total landscape area (m^2)
Radius of gyration	The mean distance (m) between each cell in the patch and the patch centroid
Cohesion	Measures the physical connectedness of the patches
Number of Disjunct Core Areas	The sum of the number of disconnected core areas contained within each patch

For each NLCD map, two main classes were created: forest and non-forest classes (Fig. 2). The forest class was comprised of the deciduous, mixed, evergreen and woody wetland NLCD classes and the non-forest class comprised of all the other NLCD classes such as developed and barren land, quarries and open water areas. With this analysis focused on forest fragmentation, the non-forest patches were treated as background values and our analysis devoted to the forest patches. A fixed edge depth, a prerequisite in the FRAGSTATS software, was defined at 30m to match the resolution of the NLCD maps.

Fig. 2: Map of Virginia showing forest and non-forest areas.
Virginia is made up of about 60 % forest areas but the degree of human activity surrounding forest areas determine species diversity and the climatic conditions within.



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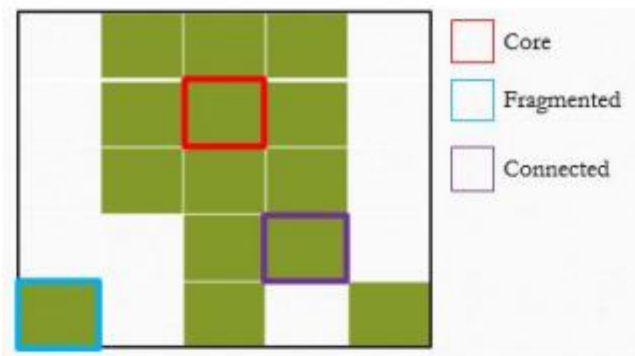
With the NLCD dataset comprising 4 main different forest types namely Deciduous Forest, Evergreen Forest, Mixed Forest and Woody Wetlands, it was important to examine which forest types were more prone to fragmentation. The NLCD Shrub class was also examined because it comprises young successional trees and forest trees less than 5 meters tall. Areas in Virginia with each of these five NLCD classes were clipped individually so that there were five raster datasets representing Deciduous, Evergreen, Mixed, and Woody Wetland forests and the Shrub class, for every time period under investigation. Consequently, these fifteen raster datasets served as inputs for FRAGSTATS to identify the difference in patch characteristics between 2001 and 2011.

For the reclassified NLCD maps showing either forest or non-forest classes, at each date (2001, 2006 and 2011), each forest patch was described by the proportion of the pixels in a surrounding neighborhood that were classified as forest pixels. Using ArcGIS, each forest pixel was then labeled as forest 'Core' if it was completely surrounded by other forest pixels. A forest pixel surrounded by less than a 100 % but more than 60 % forest pixels was labelled as 'Connected' and 'Fragmented' if the forest pixels surrounding it was less than 60 % forest pixels (Fig. 3).

Fig. 3: Forest categorization for analysis.
A forest pixel is categorized as 'Core' if it is completely surrounded by other forest pixels such as the box with the red outline. The purple outline is categorized as 'Connected' because it is surrounded by both forest and non-forest pixels but could be categorized as 'Fragmented' like the box with the blue outline if it the surrounding forest pixels were less than 60 %.

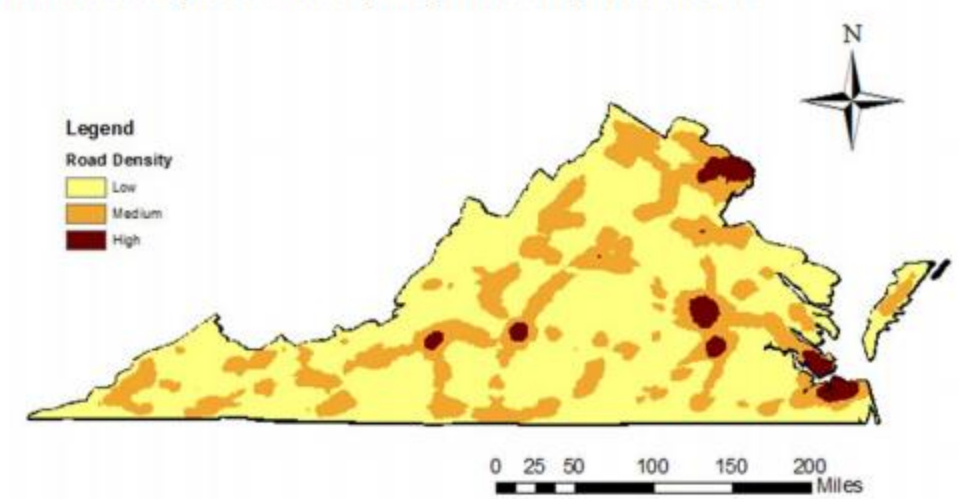


To identify contributing factors for forest fragmentation trends in Virginia, a slope map of Virginia based upon its Digital Elevation Model (DEM), was used. The slope of Virginia was then classified into 8 groups with reference to a slope report by Canada (CDA, 1974), and the amount of 'core', 'connected' and 'fragmented' forest areas in each slope class, calculated (Fig. 4). Changes in forest types across this slope gradient were then noted.

Fig. 4: Research Methodology.
The three NLCD maps were reclassified into either forest or non-forest classes before further analysis was carried out in ArcGIS and FRAGSTATS to measure forest patch characteristics.

Suspecting that the density of roads in different areas might contribute to conversion of forest areas to non-forest areas, a density map of Virginia was created based on the number of primary and secondary roads in the state. The road density map was created in ArcGIS using the Density tool and classified into three main classes based on natural breaks: High, Low and Medium Density (Fig. 5). The relationship between road density and forest fragmentation was examined by comparing the changes in the amount of 'core', 'connected' and 'fragmented' forest areas across time, in each of the three road density classes.

Fig. 5: Road density map of Virginia.
The three classes depict the number of primary and secondary roads in the area.



RESULTS

Over 60 % of Virginia is covered by forests. The *number of forest patches* differs considerably between 2001 and 2011, with an approximately 10 % increase in forest patches in 2011 compared to 2001 (Table 2). Patch density in 2001 increased from 1.7 patches per unit area to 1.88 patches in 2011.

Table 2: Patch metrics showing forest fragmentation between 2001 and 2011.
While the number of patches, total edge, LSI and NDCA have an increasing trend, the opposite is true for the patch areas.

Patch Metrics					
Year	No of Patches	Patch Area	Total Edge	LSI	NDCA
2001	175395	58.7994	594379006.7	465.6032	386029
2006	183314	56.2593	613397442.7	480.408	407312
2011	192350	53.6164	632684039.8	495.4215	426976

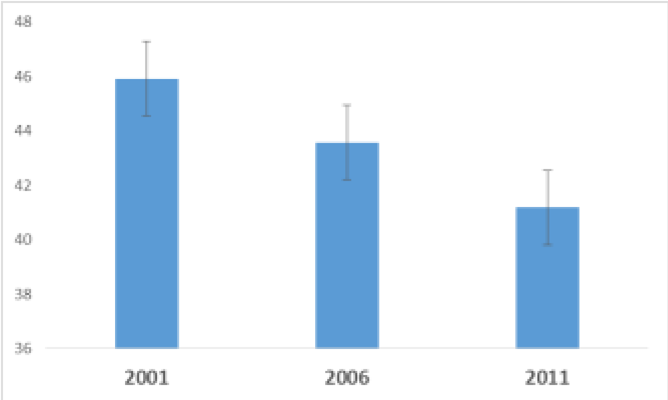
The *Number of Disjunct Core Areas* (NDCA) equals the sum of the number of disconnected core areas contained within the region. The number of disjunct core areas is an alternative to the number of patches when it makes sense to treat core areas as functionally distinct patches, instead of as part of a previously larger patch. Both NDCA and number of patches as shown in Table 2, show that, while fragmentation is increasing in Virginia with time, connectivity among forest patches is decreasing.

In FRAGSTATS, the area of a patch referred to as *mean patch size* (AREA_MN) is a measure of central tendency in the corresponding patch characteristic across the entire region, and thus equals the sum of all forest and non-forest patches. As a measure of central tendency, the mean patch size metric examines the areas of all patches within the landscape and calculates the average to represent the entire landscape. Patch size does not describe conditions within each patch but offers a primarily patch-based perspective of the landscape structure. Results in Table 2 indicate a reduction in patch area across Virginia between 2001 and 2011, pointing to the fact that Virginia's forest fragmentation continues to expand.

Total edge (TE) is an absolute measure of total edge length of a particular patch, and in this study, defines the area along the border between forest patches and the surrounding non-forest areas. Total edge is a particularly important metric for this study because forest edge areas suffer from effects from the surrounding non-forest areas and is therefore, an indication of forest fragmentation. 'Core' forest areas are those areas not affected by edge effects because they are mostly surrounded by other forest pixels. The increasing total edge length from 2001 to 2011, as shown by the results in Table 2, is a further indication of forest fragmentation. Edge density (ED), which standardizes edge to per unit area facilitates comparisons between 2001 and 2011. FRAGSTATS results show that ED increased from 57 to 61, emphasizing further, the occurrence of forest fragmentation in Virginia.

FRAGSTATS also offers the ability to specifically measure 'core' forest areas. *CORE* is a FRAGSTATS metric that represents the area in the patch greater than the specified depth-of-edge distance from the perimeter. *Mean core area per patch* (CORE_MN) therefore is the average of the core areas of the patches in an area. The results show that the core forest areas where specialist species thrive, is declining over time (Fig. 6).

Fig. 6: Core forest area percentages in 2001 decreasing over time.
This is due to increase in human activities, leading to the clearance of more and more forest areas. This has a negative impact of forest interior species that cannot thrive in forest edges.



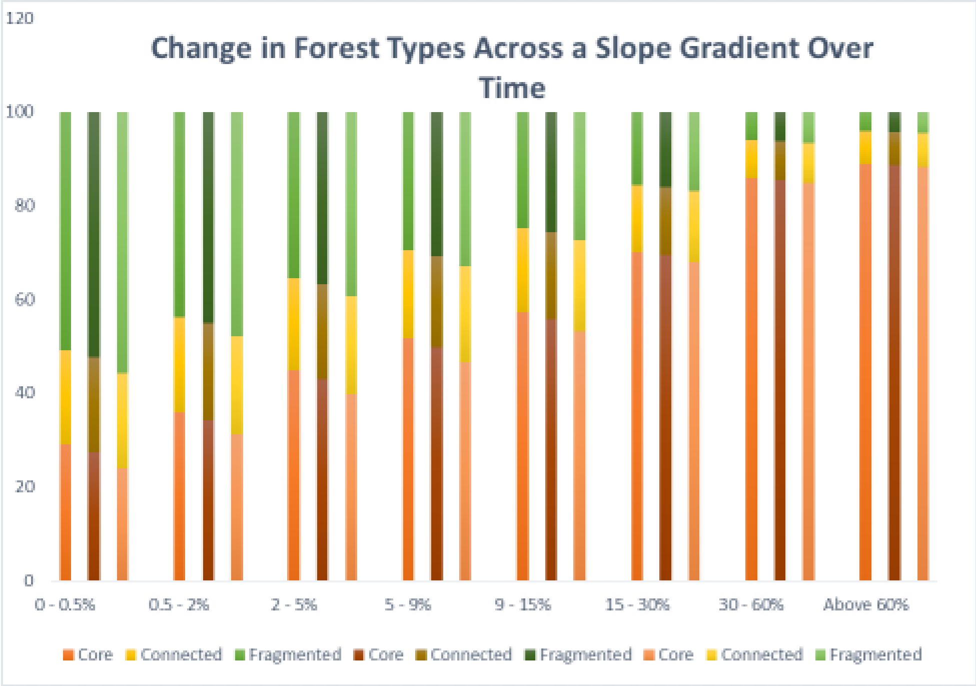
Landscape Shape Index (LSI) is a measure of the perimeter-to area ratio for the landscape as a whole. LSI can be interpreted as a measure of the overall geometric complexity of the landscape. However, it can also be interpreted as a measure of landscape disaggregation, or fragmentation. As the value of LSI increases, the more disaggregated or fragmented the landscape is. Our results in Table 2 show that fragmentation between 2001 and 2011 has been consistently increasing, while connectivity is decreasing.

Results of ArcGIS analysis indicate that, while percentages of 'core' forest areas in Virginia are decreasing significantly over time, percentages of 'connected' and 'fragmented' forest areas are increasing (Table 3). The ArcGIS results confirm the results from FRAGSTATS, which indicate that between 2001 and 2011, forest fragmentation and connectivity in Virginia have been increasing consistently.

Table 3: Percentages of forest types between 2001 and 2011.
As the percentages of core forest areas reduced from 2001 to 2011, the percentages of Connected and Fragmented forest areas increased. This is an indication of increasing human activities.

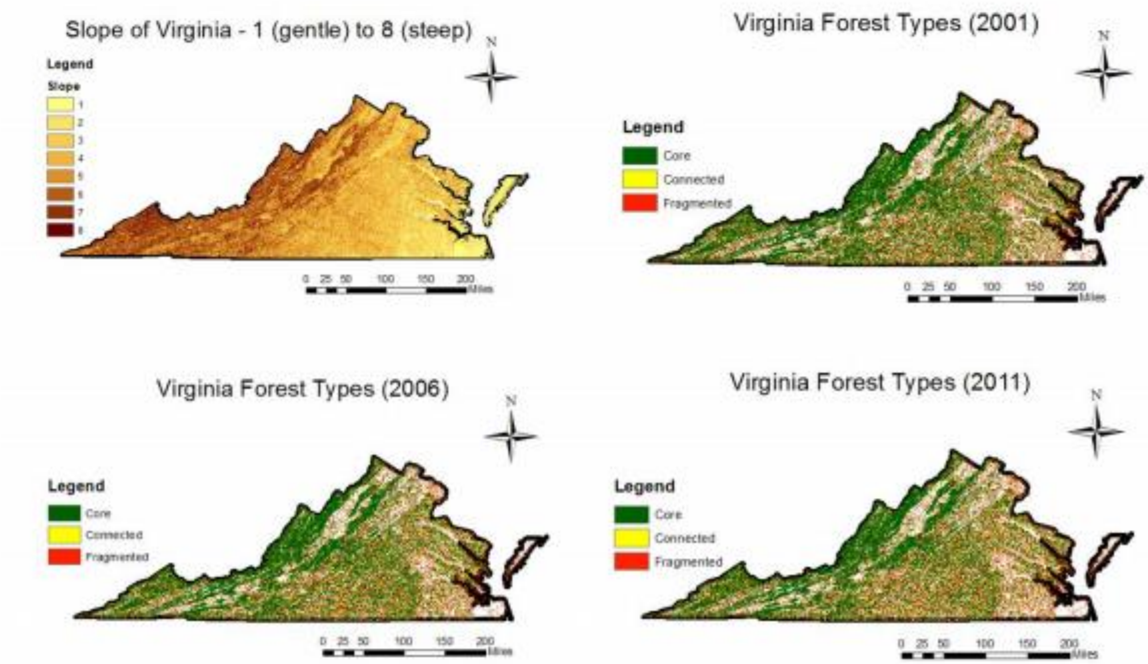
	2001	2006	2011
Core (%)	60.04	58.82	56.87
Connected (%)	15.97	16.44	16.95
Fragmented (%)	23.99	24.74	26.18

Fig. 7: Slope and forest type across time (columns in each group of three are 2001, 2006 and 2011 respectively).
Areas with steep slopes have more core forest areas, compared with areas with gentler slopes. Areas with gentle slopes have a lot more fragmented forest areas which increase over time, indicating more human activities relative to areas with steep slopes.



Our results from Figures 7 and 8 show that slope is the single most important factor contributing to this trend and that areas with relatively steeper slopes are less likely to experience changes in their 'core' forest area percentages over time. In Figure 8, inclination of core forests to steep slopes is evident as connected and fragmented forests are more pronounced in areas with gentler slopes.

Fig. 8: Map of Virginia's Topographic Slope and Forest Types.
Core forest areas are mostly in areas with steep slopes relative to areas with gentle slopes.



A closer look at the Valley and Ridge section of Virginia, where slopes are steeper, shows that core forest areas generally remained intact with only subtle changes (Fig. 9). In the Coastal Plain region where slopes are gentler, there are significant changes in the forest types of the area between 2001 and 2011 (Fig. 10). Most of the losses seen in Figure 10 occurred within core forest areas with the fragmented and connected forest areas, experiencing more gains.

The results of this study showed that areas with high road density in Virginia showed no significant change in the rates of forest fragmentation between 2001 and 2011. Our results indicate no strong correlation between forest fragmentation in Virginia between 2001 and 2011 and road density (Table 4). The percent changes in core, connected and fragmented forest areas in Virginia between 2001 and 2011, also indicated that there was no significant relationship between forest fragmentation and road density in Virginia.

Fig. 9: Valley and Ridge constitutes more core forest areas with no changes over time.
Most of the changes in forest types occur within the Piedmont and Coastal Plain areas.

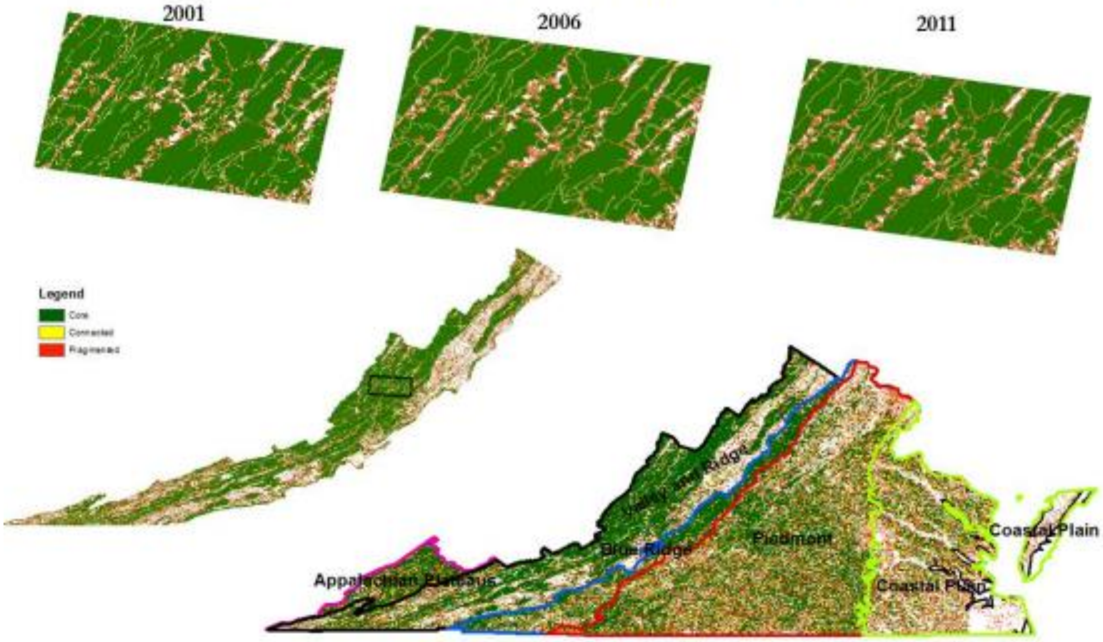


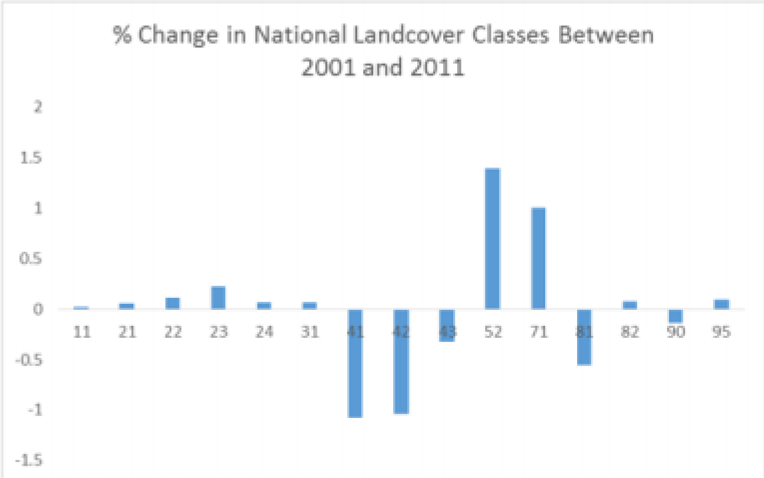
Fig. 10: Coastal Plain has gentler slopes and has more changes in forest areas over time.
This trend can be attributed to the ease and relatively low effort with which human activities such as residential construction and roads can be done in areas with gentler slopes compared to areas with steep slopes.

Table 4: Relationship between road density and forest fragmentation in Virginia.
The results show a weak relationship between forest fragmentation in Virginia between 2001 and 2011 and road density.

Road Density	% Change in Core forest area	% Change in Connected forest area	% Change in Fragmented forest area
LOW	-0.072199319	-0.07737	-0.13127
MEDIUM	0.058670343	0.009547	-0.09217
HIGH	0.085897301	0.056067	-0.04927

Between 2001 and 2011, there were changes in all the NLCD classes (Fig. 11). Remarkably, these changes were all positive except for the four forest classes and the pasture and hay class. The Shrub class which is partly forest because of its tree composition, had the most change, recording a positive 1.4 % change (Fig. 11). Negative changes in Mixed, Deciduous, Evergreen and Woody Wetland forests areas between 2001 and 2011, show that forested areas in Virginia are becoming smaller in recent years, losing to other non-forest areas such as developed lands and especially, shrub and grasslands.

Fig. 11: Changes in NLCD classes between 2001 and 2011.
All of the classes had positive changes between 2001 and 2011, except for the four forest classes and the pasture/hay class. The Shrub class had the most significant change between 2001 and 2011.



NLCD Legend
11 = Open Water
21 = Ice/Snow
22 = Developed; Low Intensity
23 = Developed; Medium Intensity
24 = Developed; High Intensity
31 = Barren land
41 = Deciduous Forest
42 = Evergreen forest
43 = Mixed Forest
52 = Shrub
71 = Grassland
81 = Pasture/Hay
82 = Cultivated Crops
90 = Woody wetlands
95 = Emergent Herbaceous Wetlands

The numbers of patches in all five NLCD forest classes in Virginia, except for the Shrub class, increased between 2001 and 2011. Within these four NLCD forest classes, the resulting patches had smaller patch sizes in 2011 compared to 2001, demonstrating a fragmentation process. The Shrub class, on the other hand, had a smaller number of patches in 2011 compared to 2001, indicating that the Virginia Department of Forestry (VDOF)'s conservation plan propelling landowners to grow more trees, is a significant factor in the increase of shrub areas in the state. With increasing patch sizes of the shrub class (Table 5), young trees in the early successional stage, are becoming more abundant, emphasizing the role of the VDOF conservation plans.

Table 5: Patch metrics showing forest fragmentation between 2001 and 2011 in the different forest type areas in Virginia.
Overall, all four forest areas experience forest fragmentation except for the Shrub forest type.

		Patch Metrics			
Forest type	**Year**	**Patch Density**	**Patch Area**	**LSI**	**NDCA**
DECIDUOUS FOREST	2001	4.1069	24.3494	855.2994	313415
	2006	4.124	24.248	861.7736	317519
	2011	4.302	23.2448	868.1741	322116
EVERGREEN FOREST	2001	25.2785	3.9559	814.5776	235430
	2006	26.4012	3.7877	830.1473	236755
	2011	28.6037	3.496	849.3857	234257
MIXED FOREST	2001	76.3368	1.31	1004.1583	127397
	2006	77.0369	1.2981	986.6209	122728
	2011	77.672	1.2875	962.4233	117043
SHRUB	2001	44.0112	2.2722	543.8879	56688
	2006	42.8946	2.3313	566.5366	64882
	2011	32.9072	3.0388	618.6549	92823
WOODY WETLANDS	2001	11.3871	8.7818	427.9522	72604
	2006	11.5459	8.6611	427.5921	72142
	2011	11.9591	8.3618	434.7146	73086

The Landscape Shape Index (LSI) and the Number of Disjunct Core Areas (NDCA), which indicate forest fragmentation and connectivity respectively, showed no clear direction. Increasing LSI values indicate increasing complexity due to fragmentation within the areas. LSI values increased between 2001 and 2011 for the Deciduous, Evergreen and Shrub forest types, decreased for Mixed forests and showed no clear pattern for the Woody Wetlands (Table 5). NDCA values generally increased for all forest types except for Mixed forests. The most significant change in NDCA values occurred between 2006 and 2011 within the Shrub class indicating that, although the number of patches decreased between 2001 and 2011, the patches became more disconnected within the same time period.

DISCUSSION

The number of patches in an NLCD layer is a measure of the extent of fragmentation of forest patches. Increases in the numbers of forest patches in Virginia over time indicates the occurrence of forest fragmentation (Magrach *et al*., 2011). As more and more patches are formed, there is increasing need to examine distances between those patches. There are many forest species that are incapable of migrating to other forest patches if those patches are separated by non-forest areas (Laurance *et al*., 1997b).

The trend of increasing number of disjunct core areas (NDCA) has significant implications for forest connectivity and for interior dwelling forest species. In light of the increasingly disturbing global climate change, many wildlife species will be forced to migrate to other areas, an activity significantly dependent on availability of migratory routes. As forest patterns determine levels of carbon sequestration, ultimately contributing to global climate change, this study is important to support policy makers as they formulate plans that examine migration patterns for wildlife species. According to a geospatial study on migratory routes between warm and cool zones, only 2 % of the eastern United States contains the connected green space needed for animals to find new homes (Coombs, 2016).

Increasing connectivity among Virginia's forested areas, is important for climate change considerations. As temperatures become warmer as a result of global climate change, connectivity among forest patches enables movement of animals, and the plant seeds they carry, to move to more habitable areas. With increasing number of disjunct core areas and number of patches in Virginia, as indicated by the results of this study, there should be concern about whether numbers of habitable areas will be sufficient to accommodate numbers of migrating species and if not, the population threshold for these species. This study therefore, informs policy makers about the nature of forest fragmentation and connectivity in Virginia and effects of increasing patchiness of forests, for wildlife movement and their population sizes.

Wegner & Merriam (1979) and Fahrig & Merriam (1985) noted that corridors between forest patches served an important function in determining population size and persistence of animals. Our results showing increases in the number of disjunct core areas means that, although numbers of forest patches in Virginia are increasing over time, forest corridors critical to link fragmented patches in order to support persistence of animal populations, are absent. Therefore, forest species in fragmented patches are often unable to interact in an effective way that ensures that their populations are viable over time.

Forest patch size determines species richness within an area, as numerous studies have proven that species richness, especially, that of birds, increases with increasing forest patch size (Fahrig, 2003; Wethered & Lawes, 2003; Dami *et al*., 2012). This effect is because, external factors such as wind intensity and light penetration have greater influence on smaller forest patches (Honnay *et al*., 1997). As forest patches in Virginia decrease in size over time, it is reasonable to expect a consequent decrease in forest species persistence, especially bird species (Magrach *et al*., 2011). Decreases in area of the forest patches force a decrease in the number of species in each forest patch. This poses a problem for a lot of forest species since Magrach *et al*. (2011) showed small populations to be more susceptible to stochastic extinctions and/or allee effects (a phenomenon describing correlations between population size and mean individual fitness). Hence, results from this study showing decreasing forest patch sizes, need to be corroborated with Virginia's population sizes of certain species, such as ovenbird (*Seiurus aurocapilla*) and wood thrush (*Hylocichla mustelina*), to either confirm earlier research or provide alternative findings.

Increases in forest edge areas in Virginia, as seen from results of this study, has several implications because microclimatic conditions in forest edge areas can be significantly different from those in core forest areas, and can therefore, create either favorable or adverse conditions for plant growth and consequently, animal populations (Smith *et al*., 2018). Light, temperature, moisture content and wind intensity are some of the environmental differences between core forest and forest edge areas (Ritter *et al*., 2005). Increasing forest edge areas makes the region more vulnerable to fires through desiccation of fuels and greater exposure to potential human ignition sources (Laurance & Curran 2008).

Matlack (1993) noted that increased incident solar radiation is the singular most important factor differentiating the microenvironment of forest edges and core forest areas. With the results of this research showing a reduction in core forest areas, forest species that thrive best in darker areas are more prone to extinction, giving way to those species that are more suitable for brighter areas. Thus, forest areas in Virginia are likely to experience changes in both plant and animal populations over the coming years if steps are not taken to curb forest fragmentation.

Kautz *et al*. (2013) also noted that forest edge areas are more predisposed to species invasion and pest infestations. For example, the Ebola disease outbreak in Africa between 2013 and 2015 is purported to be a result of deforestation and fragmentation (Wallace *et al*., 2014; Bausch & Schwarz, 2014). Rulli *et al*. (2017) showed that centers for first infection of the Ebola disease were in areas with high degrees of forest fragmentation. As a result, more humans come into contact with disease reservoirs as forests are cleared for industrial and residential constructions, exposing humans to zoonotic infections (Rulli *et al.,* 2017). In the United States, spread of Lyme disease in the mid-70s, can be directly traced to anthropogenic deforestation that occurred during the expansion of the American suburb (McGrath, 2014). Proximity of humans to deer populations infected with the *Borellia burgdorferi* bacteria passed on by the bite of a deer tick, resulted in approximately half a million cases of Lyme disease infections in 2013 (McGrath, 2014). Because one of the most consequential effects of forest conversion is increased forest edge area (Smail & Lewis, 2009), it is important for policy stakeholders to investigate effects of increasing forest edge areas and decreasing core areas in Virginia, on the prevention of disease outbreaks that affect human population and health. Hence, information on forested areas that have been converted to non-forest areas, the basis of our study, is crucial for identifying hotspots for potential disease outbreaks.

The complexity of a landscape determines population sizes of many forest species. Rosch *et al*. (2013) noticed that populations of generalist insect species such as the leafhopper (Cicadellidae), decreased with increasing forest patch isolation but increased within complex landscapes. This effect is because, in simple landscapes, species such as the leafhopper may find it difficult to reach the next suitable site, unable to find suitable alternative resources or habitats with a similar vegetation type or structure during dispersal (Rosch *et al.,* 2013). More complex landscape structures provide more suitable habitats for these generalist species. With our results showing an increase in landscape complexity in more recent years, it is important to understand what this means for both generalist and specialist forest species in Virginia.

Reduction in core forest areas in Virginia is likely to pose a problem for forest interior species because their population sizes will be significantly affected. On the other hand, increased connectivity and increased edge areas as a result of more 'fragmented' forest areas, will increase the population sizes of species that thrive at forest edge areas. This means that changes in core, connected and fragmented forest areas over time, can lead to changes in forest composition (Smail & Lewis, 2009). Changes in forest composition have direct impacts on the complexity (as noticed in this study) and stability of forest ecosystems (Bodin & Wiman, 2007; Pentilla *et al*. 2006; Quetier *et al*. 2007). Holway (2005) showed that changes in forest compositions, and more specifically, decrease in core areas, led to the introduction of nonnative and invasive species which may have unknown deleterious consequences. For instance, the spread of the emerald ash borer (*Agrilus planipennis Fairmaire*) in the Great Lakes region of the United States led to the destruction and near-extinction of some ash tree species (Smail & Lewis, 2009).

The extent of forest fragmentation and connectivity in an area has different consequences for species living in that area. Hence, an analysis of forest fragmentation that takes into consideration use of multiple scales of resolution is more likely to identify suitable habitats for primary species in the area and provide insight on how corridors will be useful for all species present. With growing demands for strategies that allow for forest fragmentation estimates across multiple scales, it is important for future studies to consider the risk of biased results from fragmentation metric comparisons across different scales.

Losses in core forest area and simultaneous gains in fragmented and connected forest areas highlight the fact that slope has a strong control on urban development and forest fragmentation. Urban development is typically limited by high costs, low commercial value, and risk of landslides; all these are pronounced in areas with steep slopes (Gao & Yu, 2014). Because steep slopes cannot be used to build large urban patches, variation of slope within an area, further fragments forest areas.

Our result showing correlations of slope and forest fragmentation, is consistent with studies that have shown that land areas with steeper slopes have higher production and transportation costs and are therefore less likely to be converted from natural to anthropogenic land cover (Berry *et al*., 1990; Sheppard & Barnes, 1990; Wickham *et al*., 2000). This result is consistent with findings from Echeverria *et al*. (2008) who showed that, between 1976 and 1999 in southern Chile, forest fragmentation occurred mainly at edges of small fragments situated on gentle slopes (less than 10°). With forested areas that have steeper slopes with more core forest land areas and fewer changes between 'core', 'fragmented' and 'connected' forest types over the ten year time period, it is important for policy makers in Virginia to concentrate their efforts in areas with gentler slopes because these areas are subject to forest fragmentation.

van Kooten & Folmer's (2004) theories on land rent suggesting that land use is allocated to maximize revenue from a land of a specified quality, provides insight to why this is so. With forest fragmentation being a result of human activities and a direct consequence of human land use decisions, costs of developing a unit of land plays a significant role. Land areas with gentler slopes are more easily developed as material transportation costs are cheaper and are generally, more suitable for agricultural purposes (Alig *et al*., 2005).

Given the knowledge that increasing human activities correlate with increasing forest fragmentation and that human activities are noticeably higher in areas with high road density, there should be an increasing rate of forest fragmentation in areas with higher road density. Roads have been found to promote forest fragmentation by dividing large forest patches into smaller areas and in so doing, creating more forest edge areas at the expense of core forest areas. For instance, Reed *et al*. (1996) showed that roads contributed significantly to forest fragmentation in the Rocky Mountains. However, Miller (1994) indicated that road density in an area could be very high or increasing without increasing the number of forest patches or forest edge areas. Our results showing no correlation between road density and forest fragmentation suggest that our study period between 2001 and 2011 might not be enough time to fully capture the effect of road on forest fragmentation and therefore, subsequent studies should examine a longer time period to fully understand the impact of roads. The insignificant correlation between road density and forest fragmentation patterns in Virginia also point to the fact that other factors have more substantial impact on the forest fragmentation process in Virginia.

This led to the exploration of the different forest types in Virginia to identify how more susceptible some forest types are to forest fragmentation, compared with others. The diminishing trend of core forest areas and the increasing trend of connected and fragmented forest areas in Virginia between 2001 and 2011, raises questions on the specific types of forests undergoing conversions. For instance, pine trees that occupy approximately 20 % of Virginia's forest (VDOF, 2014), are undergoing significant harvesting followed by replacement with other species. As a result of this, pine trees in Virginia are currently considered as diminished species, triggering the Virginia Department of Forestry to initiate a pine tree restoration program with cost-share plans to enable interested landowners to re-establish them (Gagnon, 2016). It is therefore important to know the specific types of forests undergoing fragmentation in Virginia.

Shrub lands, as defined by the NLCD, are areas with trees that are less than 5 meters tall and dominated by more than 20 % shrub canopy. This NLCD class comprises true shrubs, young trees in an early successional stage or trees stunted from environmental conditions. Going by this definition, this class will comprise all the areas that undergo afforestation and reforestation annually. Given the Virginia Department of Forestry's Conservation Incentive Programs that were rolled out in the late 1990s, allowing Virginia land owners to receive financial benefits for helping in conservation efforts, it is not surprising that the NLCD Shrub class had the most significant change within the study time period (Fig. 10).

A closer look at the five NLCD forest classes to examine the extent of forest fragmentation between 2001 and 2011, concentrates on metrics that serve as the most significant indicators of forest fragmentation. The process of forest fragmentation has three main indicators namely, a) increase in forest patches, b) decrease in forest patch areas and c) increasing isolation between the resulting forest patches. The four patch metrics in Table 5 were examined to identify the extent of forest fragmentation in specific forest types in Virginia between 2001 and 2011.

Table 5 confirms the report of Heilman *et al.* (2002) that, use of the number of forest patches as an indicator of forest fragmentation should always be corroborated with the connectivity between the patches before conclusions are made. The process of division of large forest patches into many smaller patches is not always an ecological negative (Heilman *et al.*, 2002). In some forests with natural patchiness that allows for connectivity between those patches, forest fragmentation becomes a sign of higher ecological integrity (Heilman *et al.,* 2002). For the Mixed forest class where the number of forest patches is increasing over time but the disconnectivity (LSI) is decreasing over the same period of time, the disturbance could be said to be more natural, such as fires or localized windthrow (Heilman *et al.*, 2002), rather than anthropogenic disturbances. The reverse is also true, an example being the Shrub class, which is becoming disconnected over time although the number of patches is decreasing over the same period of time. This follows that the Shrub class is manipulated by anthropogenic activities rather than more natural procedures. This result highlights the role of the VDOF in forest conservation and the implications of management decisions. The results of this study provide resource managers and policy makers the information required to guide forest management decisions.

Conclusion

Results from FRAGSTATS present a clear description and hence, are useful for quantifying forest fragmentation. Results from the analysis show significant differences in forest fragmentation in Virginia from 2001 to 2011 with trends indicating increasing forest fragmentation and connectivity. This information is useful in developing appropriate forest management strategies that aim to mitigate adverse impacts of land use changes, especially, those due to human activities.

The reshaping of forest boundaries, changes in forest patch sizes, reduction in core forest areas and the consequent increase in total forest edge areas, increase the risk of zoonotic infections with important impacts on human health worldwide. The trend and extent of forest fragmentation and connectivity in Virginia have crucial externalities associated with human health, all of which should be accounted for while evaluating the costs, risks, and benefits of changes made to forests in Virginia.

Results also show that the slope of an area significantly contributes to the degree of forest fragmentation in that area as human activities in steeper sloped areas, reduce. Our study therefore presents an important framework for development in Virginia as policy makers can use this as a tool for making decisions. With the observed strong relationship between land use and slope in Virginia, subsequent research on what these changes mean to the populations of biodiversity present in Virginia will provide policy makers with relevant information required for conservation and development purposes. Careful consideration of slope categorization in future studies will help identify the underlying contributions of slope to landscape uses and avoid arbitrary boundaries.

Given the decreasing number of patches but increasing connectivity within the Shrub class which comprises young trees in the early succession stage, it is important for future studies to more fully consider the role of management decisions on forest conservation in Virginia. Anthropogenic interferences might have positive implications depending on the scale of measurement being used. In determining the efficiency of management decisions, it is important that every factor be fully assessed and the collective impact, recognized.

Lastly, some studies have shown better fits for regressions when soil properties are used as explanatory variables for forest fragmentation studies. Results from these studies show that significant variation in soil properties within a specific area will result in heavier forest fragmentation. Since Virginia is not a spatially homogenous unit, it is important for future studies to consider the role of soil properties on forest fragmentation and connectivity.

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Appendix B

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FOREST FRAGMENTATION ANALYSIS FROM MULTIPLE IMAGING FORMATS

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ABSTRACT

In landscape ecology, forest fragmentation studies with emphasis on effects of scale on fragmentation patch metrics, is an important research area. With increasing availability of satellite data at multiple scales and varied resolutions, it has become important to understand effects of comparing fragmentation metrics acquired from coarse resolution images and those from finer resolution imagery. This is crucial because coarse resolution images such as Landsat imagery, are relatively easier to find because of their cheaper costs, availability and broad coverage, whereas finer resolution imagery is more expensive and therefore, spans only small areas. This paper examines effects of varied spatial resolutions on common fragmentation metrics using Landsat, Sentinel, National Agricultural Imagery Program (NAIP) and Unmanned Aerial Vehicle (UAV) imagery obtained in November, 2017 of the Whitethorne area near Blacksburg, Virginia. The images are analyzed using FRAGSTATS and ArcGIS software programs. The results show significant differences in fragmentation metrics despite simultaneous acquisition of all images in the same area. Discussion of results obtained in this study centers on the reasons for this disparity, and examines uses of imagery of different resolutions for forest fragmentation analysis.

Keywords: Forest Fragmentation, Landsat, UAV, forest patches, spatial resolution, Patch metrics

INTRODUCTION

Forest fragmentation, one of the major threats to biodiversity and forest conservation, is the process through which formerly large and continuous forest areas are converted to small, isolated patches (Haila, 1999; Loyn & McAlpine, 2001). Reduction in sizes (areas) of remaining forest patches, increased isolation and loss in connectivity; and increased edge effects are the three main consequences of forest fragmentation (Saunders *et al.*, 1991; Forman, 1995). Thus, forest fragmentation indices have the capacity to serve as spatial indicators for assessing health of forest ecosystems and are commonly considered biodiversity indicators in national forest inventories (Soledad & Saura, 2005). Forest fragmentation indices are important for assessing whether critical components and functions of forests are being maintained over time (Soledad & Saura, 2005).

The purpose of forest fragmentation analysis is to allow users to visualize and quantify the extent of forest fragmentation while tracking changes in fragmentation and connectivity over time (Riitters *et al.*, 2000). Research conducted by Riitters *et al.* (2000) which forms the basis of most forest fragmentation work, was originally developed to assess forest fragmentation at the global level using 1-km land cover information premised on the use of image convolution where a fixed area, roving 'analysis window', is centered over a forest pixel identified by a raster land cover map.

In remote sensing, one of the most widely used processes involves image classification. Image classification is the process of converting the information in an image based on the spectral response of the Earth's surface, into a thematic map that shows several classes of interest (Foody, 2008). In order to measure the accuracy of resulting thematic maps from the image classification process, it is necessary for users of these maps to evaluate their quality. The process of measuring the quality of classified thematic maps is referred to as Image Classification Accuracy Assessment, shown to be a difficult variable to assess because of problems associated with class discrimination and the spatial resolution of the images used in the classification process (Foody, 2008; Pontius & Cheuk, 2006; Lu & Weng, 2007).

Spatial resolution of the input land cover information is one of two most significant considerations in forest fragmentation analysis; the second being the desired width of the forest edge (Hurd & Civco, 2008). In forest fragmentation analysis, both of these two considerations are related and play significant roles in remote sensing, where images are analyzed and classified in order to map forest patches. Remote sensing analysis of forest fragmentation is very sensitive to scale of the maps used. With the availability of remote sensing data at varying spatial scales, a primary concern in fragmentation analysis is in defining an appropriate spatial resolution that ensures that results represent good ecosystem indicators (Lausch & Herzog, 2002).

In analyzing effects of spatial scale on landscape pattern indices, Saura (2004), found lower fragmentation at coarser spatial resolutions. Results from Saura (2004) contradicts that of Garcıa-Gigorro & Saura (2005) who concluded that images with finer spatial resolution underestimated forest fragmentation and reasoned that the utility of finer resolution images for forest fragmentation analysis is probably overestimated. Other studies have also shown that despite the usefulness of high-resolution imagery in capturing small habitat patches compared with lower-resolution imagery, high resolution imagery has the disadvantage of producing more canopy shadow and complicates processing and comparisons of multiple images (Masouka *et al*., 2003; Kennedy, 2009; Asner & Warner, 2003).

Characteristics of sensors do not only affect levels of image detail (spatial resolution) but also the radiometric resolution (the sensitivity of the sensors to detect differences in reflected or emitted energy (Narayanan *et al*., 2002). This means that the brightness of remote sensing imagery is dependent on sensors used to record electromagnetic energy of the objects in the scene (Narayanan *et al*., 2002). For instance, while the Landsat MSS has a radiometric resolution of 6 bits, the Landsat ETM+ has a radiometric resolution of 9 bits which emphasizes differences between agricultural and forest covers despite the small differences in their reflected energy.

In this study, we aim to determine effects of spatial resolution of remote sensing images on calculation of landscape metrics commonly used in forest fragmentation studies. Our study area in Virginia, where landscapes are heterogeneous in nature and rates of development, determined by human activities, have resulted in significant landscape changes, is ideal. This study differs from previous studies such as Wickham & Riitters (1995), Frohn (1998) and Wu *et al*. (2002) who concentrated solely on effects of spatial resolution on landscape indices devoid of the consideration of the dates of acquisition of the satellite images. In this study, we directly compare fragmentation indices on simultaneously acquired satellite images of different spatial resolutions for the same landscape. Hence, it offers a better understanding of the effect of spatial resolution on forest fragmentation and connectivity analyses.

Despite knowledge of significant differences in fragmentation metric values from images of different spatial resolution, effects of spatial resolution on fragmentation metrics is not fully understood. Forest fragmentation studies such as those of McGarigal & Marks (1995) have cautioned against comparison of fragmentation metrics obtained from images of varying spatial resolutions. This lack of comparability limits the importance of quantitative forest fragmentation analysis (Saura, 2004). This study therefore provides further insight into effects of spatial resolution of different satellite imagery on forest fragmentation metrics and identifying those metrics that can be compared across differing spatial resolutions.

In remote sensing, spatial resolution is important for determining levels of detail obtained from an area. Satellite imagery with high spatial resolution has produced more accurate estimates, where the accuracy of their classifications, have been assessed (Geza & McCray, 2008; Lin *et al*., 2010; Boyle *et al*., 2016). However, high resolution imagery although mostly beneficial because of the level of detail it affords, the issue of shadows, a nuisance that obscures important details, is more compounded in high resolution imagery. Because different services require different spatial resolutions, it is important for remote sensing research to identify the most appropriate resolution for specific objectives, given the classes of interest to be classified, in order to save both time and money. This research therefore, highlights advantages and disadvantages of satellite imagery of various resolutions for forest fragmentation analyses. Knowledge of what satellite imagery to use for what analysis is of particular value to scientific researchers and institutions that collect remote sensing data for forestry inventory collection and management.

METHODS

Study Area

Virginia, surrounded by the states of Maryland, West Virginia, Tennessee, Kentucky and North Carolina, has a population of approximately 8.5 million and occupies an approximate area of 42,775 square miles. Virginia includes major cities such as Norfolk, Chesapeake, Newport, and Richmond, its capital. Oak hickory is the most common forest type in Virginia accounting for about 61 % of the forested land (Rose, 2015; Virginia Department of Forest (VDOF), 2016). While the most productive sites in Virginia have northern red (*Quercus rubra*) and white oak (*Quercus alba*), mockernut hickory (*Carya tomentosa*) and pignut hickory (*Carya glabra*), the less productive sites in southwest Virginia, have mostly chestnut (*Castanea*) and scarlet oak (*Quercus coccinea*) trees (Gagnon, 2016). Pine trees account for approximately 20 percent of Virginia's forest cover with native pine species like the longleaf (*Pinus palustris*) and shortleaf (*Pinus echinata*), dominating in these forests. The remaining 20 % of forest areas in Virginia comprises oak-pine forest types. Throughout Virginia, especially in low-lying wet areas, Bottomland hardwoods, stable since 2001, make up approximately 5 % of forests (Gagnon, 2016). Bottomland hardwood forests have a lot of tree diversity including swamp chestnut (*Quercus michauxiik*), cherrybark oak (*Quercus pagoda*) and American sycamore (*Platanus occidentalis*). State forests in Virginia are only about 0.5 % of the total forest in Virginia while over 80 % of the forests are privately owned and managed (VDOF, 2014).

Many factors such as population growth rate, influence the quantity, quality and sustainability of forest resources in Virginia. In 1992 when the Virginia Department of Forestry (VDOF) performed a Forest Resource Assessment using GIS analysis, the result showed increasing fragmentation of its forest areas as a result of commercial and residential development. The trend of forest fragmentation in Virginia continues as confirmed by a recent study assessing the extent of forest fragmentation between 2001 and 2011 (Fynn *et al.*, 2018).

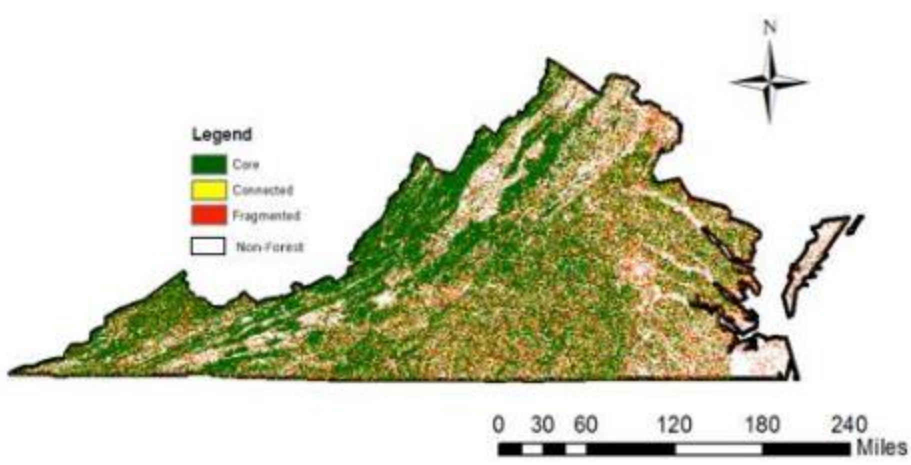
Stemming from concern for forest resources in Virginia, VDOF, using previously collected data in their Forest Inventory Analysis (FIA) pool, in conjunction with 2000 Landsat TM satellite imagery, classified the Virginia landscape into forest, non-forest, and water. The output of this analysis has been used in forest resource assessments, forest fire risk modeling, water quality management, fragmentation analyses, forest economics, and conservation efforts (VDOF).

With the widespread use of Landsat data at 30 meter resolution for forest analyses in Virginia, the concern for many ecologists lies in whether conservation efforts will be more useful if finer resolution satellite imagery is used. For instance, missed detection of forest edges will lead to false conclusions about the real status of forests, with some forests identified as intact and therefore, not receiving the needed attention even though they may have experienced disturbances. Overall, Virginia will benefit from improved information derived from satellite imagery, given that it has forest areas with varying states of disturbances (Figure 1).

Within Virginia, it was important to identify an area that has undergone a lot of anthropogenic changes and therefore, has lost a great percentage of its original habitat, in order to accurately capture effects of satellite imagery resolution on forest fragmentation studies. Myers *et al.* (2000) explain that in order for conservation efforts to be effective, a promising approach is to identify 'hotspots', or areas featuring exceptional concentrations of endemic species and experiencing exceptional loss of original habitat.

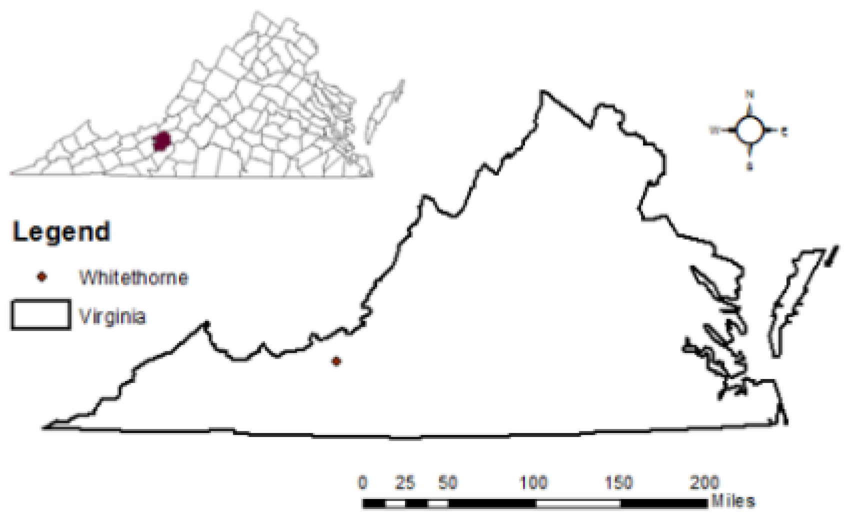
Fig. 1: A map of Virginia showing three types of forests: Core, Connected and Fragmented.

Most of the Core forest areas are in the Appalachian Plateau, Valley and Ridge and Blue Ridge physiographic regions.



Whitethorne, located near Blacksburg, within Montgomery County, Virginia, includes parts of the New River and Toms Creek (Figure 2). It includes forested and agricultural areas, interspersed with residential constructions. Previously, with a mainly forest outlook, urbanization of the Montgomery County due to the expansion of Virginia Tech, has resulted in the establishment of residential facilities and agricultural lands. The Whitethorne area was selected for this study because of the increasing human population in the area and resulting increases in forest fragmentation. Changes in the land use patterns of the Whitethorne region over the years, make the area conducive for forest fragmentation studies.

Fig. 2: A map of Virginia showing the Whitethorne region.
The map of Virginia above shows Montgomery County where Whitethorne is located.



Spatial data

To examine the Whitethorne landscape, we obtained cloud-free geometrically corrected satellite scenes for the region. These included a four band (red, blue, green and infrared) Landsat Thematic Mapper (TM) scenes (30 meter resolution) acquired on 15th November 2017 from the United States Geological Survey (USGS) archive, Sentinel imagery acquired on 18th November 2017 (Sentinel 2; only 10 and 20m resolution bands were used) from the Scientific Hub, National Agriculture Imagery Program (NAIP) imagery dated 30th November, 2017 (0.25 meter resolution) from the United States Department of Agriculture (USDA) Farm Service Agency databases, and Unmanned Aerial Vehicle (UAV) imagery obtained on 8th November, 2017 (0.03 meter resolution) (Table 1).

Landsat, Sentinel and NAIP imagery were used because they are readily and freely available and represent commonly available satellite imagery used for forest fragmentation analyses. NAIP imagery, administered through the United States Department of Agriculture's Farm Service Agency, comprising of red, blue, green and near infrared bands, is made up of individual image tiles with each tile based on a 3.75-minute longitude by 3.75-minute latitude quarter quadrangle plus a 300-meter buffer on all four sides. Dates of acquisition for all data used were close in time to ensure consistency in phenology and in vegetation states. UAV imagery used for the analysis was however not readily available and was scheduled and collected personally using a Sequoia_4.9_4608 x 3456 camera model with RGB features. The Average Ground Sampling Distance (GSD) for the UAV imagery was 3.37 cm /1.32 in and an approximate processing time of 5 hours was used on 1221 geolocated images.

Table 1: Satellite imagery used for analysis.
All the images were acquired in November to ensure more effective comparisons devoid of vegetation phenological differences due to time of year.

Satellite Image	Acquisition Date	Spatial Resolution (meters)
Landsat	11/15/2017	30 × 30
Thematic Mapper		
Sentinel 2	11/18/2017	
Band 2		10 × 10
Band 3		10 × 10
Band 4		10 × 10
Band 5		20 × 10
Band 6		20 × 10
Band 7		20 × 20
Band 8		10 × 10
Band 8A		20 × 20
Band 11		20 × 20
Band 12		20 × 20
NAIP	11/30/2017	0.25 × 0.25
Unmanned Aerial Vehicle (UAV)	11/08/2017	0.03 × 0.03

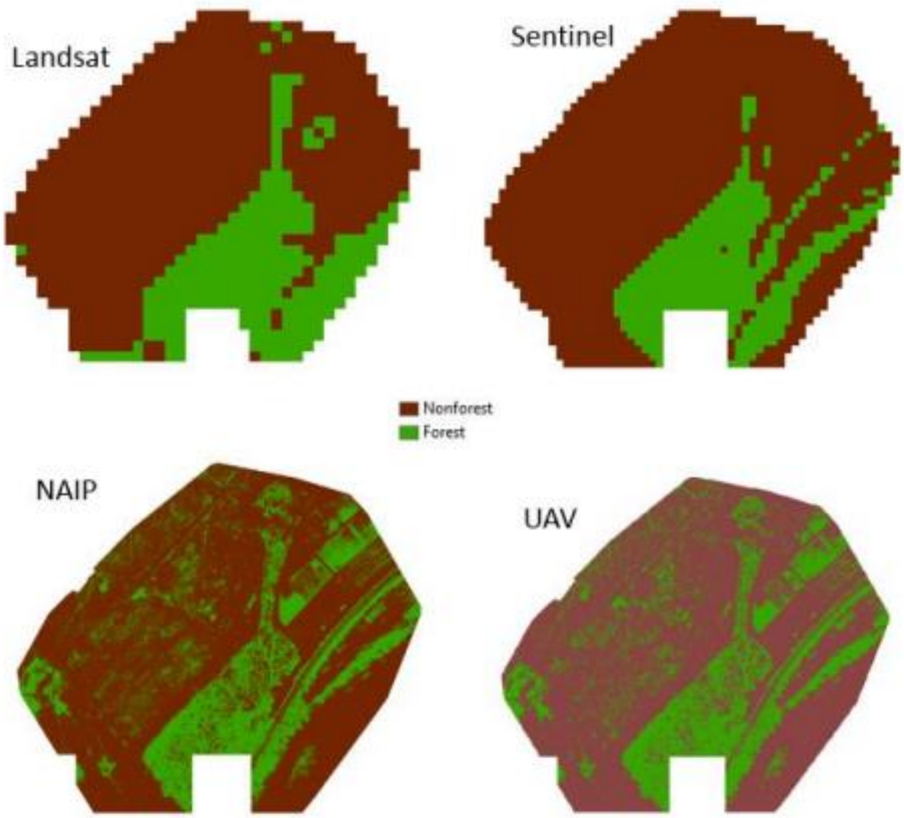
Methodology

Across the study area, 60 points were randomly selected to serve as training samples for classification. These 60 random points were field-verified on-site to confirm that their classification as either forest or non-forest, is accurate. Using ArcGIS, we assigned each point to a land cover class; forest or non-forest. These 60 locations were used for classification of Landsat, Sentinel, NAIP and UAV images, by selecting training data polygons around the points, with the same reflectance characteristics, as the point location. Classification accuracy was assessed for all images based on 500 stratified random points.

ArcGIS was used to conduct a supervised classification of the images. The 60 training data locations used to generate signature files were used to classify each of the images into either forested or non-forested areas. Pixels in each image were compared numerically and algorithmically with the training data by the ArcGIS software via the 'Maximum Likelihood' tool, to allocate the two classes.

Raster outputs from the classification of the 4 images (Figure 3) were used as inputs in FRAGSTATS to calculate patch metrics. In ArcGIS, all the non-forest areas were reclassified to have NODATA values before the raster images were analyzed in FRAGSTATS. This change was done so that, patch metrics calculated in FRAGSTATS, reflected only the forest regions in the raster images.

Fig. 3: Classified raster images of the satellite imagery analyzed in FRAGSTATS. Between the NAIP and UAV imagery, there is only little difference visually but the number of pixels in the two classes are significantly different.



Patch metrics such as patch density, total edge distance, perimeter-area ratio and shape indices were calculated for each of the images (Table 2). The Patch Density metric has the same basic utility as number of patches as an index, except that it is more effective for comparison because it expresses the number of patches on a per unit area basis. Patch Density is calculated by dividing the number of patches by the total landscape area and is useful in determining the number of subpopulations in a spatially-dispersed population for species exclusively associated with that habitat type e.g. forests. Patch density in FRAGSTATS is greater than 0 and has no maximum limit.

Table 2: Patch metrics calculated in FRAGSTATS (McGarigal and Marks, 1995).
These patch metrics are important indicators of fragmentation and connectivity

Patch Metric	Description
Patch Density	The number of patches divided by total landscape area measured per 100 hectares. • Images with high patch density indicates a higher number of patches identified and is thus, considered to be more fragmented.
Largest Patch Index	The area of the largest patch divided by the total landscape area, multiplied by 100 (to convert to a percentage)
Edge Density	The sum of the lengths (m) of all edge segments, divided by the total landscape area (m^2), converted to reflect every 100 hectares.
Landscape Shape Index	The sum of the landscape boundary and all edge segments (m) within the landscape boundary, divided by the square root of the total landscape area
Radius of gyration	The mean distance between each cell in the patch and the patch centroid
Cohesion	Measures the physical connectedness of the patches

The Largest Patch Index (LPI) metric quantifies the percentage of the total landscape comprised by the largest patch. It is therefore a measure of dominance showing the degree of variability within the landscape (Vizzari & Sigura, 2013). This metric identifies the largest forest patch within a specific landscape and therefore, determines the health of species with respect to competition and interactions between species since the size of patches have a direct impact on species population dynamics. LPI is different from the Landscape Shape Index (LSI) which is a measure of aggregation, measuring perimeter-area ratio for the landscape, calculated by dividing the total length of patch edges by the minimum measured edge length.

The Edge Density (ED) metric reports total patch edge length within a landscape on a per unit area basis. It equals the sum of the lengths, in meters, of all edge segments in the landscape, divided by the total landscape area. In this study, edge areas is defined as the area 300ft away from a patch boundary. Radius of gyration (a patch metric affected by both patch size and patch aggregation or connectivity), was also measured in meters for each image by calculating the mean distance between each cell in a patch and the patch centroid. Radius of Gyration is a measure of how far across the landscape a patch extends its reach.

The patch Cohesion metric measures physical connectedness of the patch type in the landscape under study. Cohesion increases as patches in the landscape become more aggregated and has a range between 0 and 100 in FRAGSTATS. Total Core Area (TCA) (a measure of the aggregation of the core areas in each patch), was also measured for each image. Metric results for each of the four images were subsequently compared, to note any differences.

Results and Discussion

Considerable differences were found between the metric values of Landsat, Sentinel, NAIP and UAV images from FRAGSTATS. Contrary to our initial hypothesis that differences between the four images, might be subtle, if any, the case is not so for fragmentation metrics. Fragmentation metrics such as number of patches or patch density, Largest Patch Index, Landscape Shape Index, Edge Density and Radius of Gyration, showed similarities between Landsat and Sentinel Imagery compared with NAIP and UAV images (Table 3). The similarity of results of Landsat and Sentinel and then, NAIP and UAV, were to be expected because Landsat and Sentinel sensors have similar spatial resolutions compared with NAIP and UAV imagery.

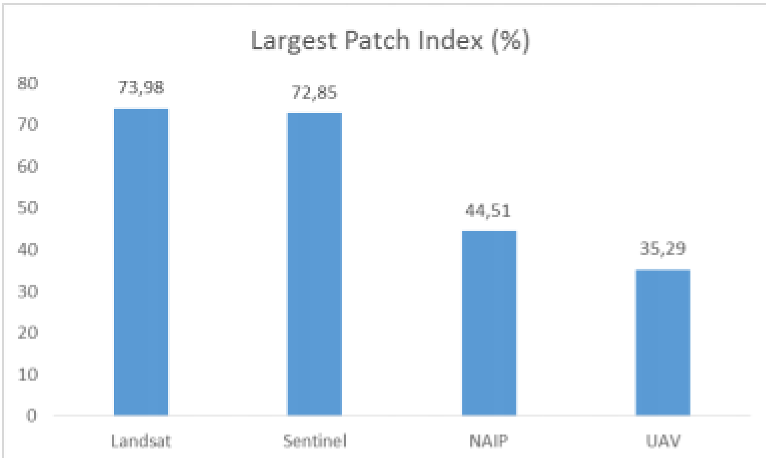
Table 3: Patch Metric Values from FRAGSTATS.
There is a lot of similarity between metric values of Landsat and Sentinel and another cluster of similar values for NAIP and UAV metric values. This trend is because of comparable spatial resolutions

PATCH METRICS					
IMAGE	**Patch Density (patches/100ha)**	**Largest Patch Index (%)**	**Edge Density (per 100ha)**	**Landscape Shape Index**	**Radius of Gyration (m)**
Landsat	0.10	73.98	0.55	2.52	84.0259
Sentinel	0.14	72.85	0.69	2.86	71.4634
NAIP	734.37	44.51	259.99	77.93	0.093
UAV	1321.66	35.29	346.97	120.96	0.091

Patch Density affects the stability of species interactions and opportunities for coexistence among different species. Patch Density, is however, constrained by the spatial resolution of the image used, since it is measured when every cell is a separate patch. This constraint is evident in the results reported in Table 3, as Patch Density differed considerably depending on the spatial resolution of the images used. Careful consideration of spatial resolution is important because if Patch Density is used as the only metric of comparison between images of varying spatial resolution, investigators will reach erroneous conclusions, given the differences in results reported in Table 3. This means that in a given area, depending on the spatial resolution of the satellite imagery used for the analysis, patch density value can either be lower or higher than the actual ground information. Hence, it is important for conservationists to consider the spatial resolution of images used in a fragmentation analysis before making decisions based only on Patch Density.

The significant differences between LPI metric values for the four images was not expected since this metric is thought to be independent of spatial resolution (Aithal *et al.*, 2012). Our results, however, show that, images with lower spatial resolution (such as Landsat and Sentinel) have the tendency of skipping small patches within larger patches and therefore aggregating smaller patches as individual larger patches. With increasingly finer spatial resolution, the distinction between smaller patches is more easily recognizable, as is seen in Table 1 and therefore, LPI declines (Figure 4). This result serves as an important factor for conservationists and policy makers as it shows the value of using higher resolution imagery.

Fig. 4: Largest Patch Index (LPI) reduces with increasing spatial resolution.
Although the same area at the same time is analyzed, the spatial resolution of the satellite imagery used, results in differences in the calculations of the LPI. There is a significant reduction in LPI values obtained from fine resolution imagery (NAIP and UAV) compared with coarse resolution imagery (Landsat and Sentinel).



Edge Density, like Patch Density, is very sensitive to the spatial resolution of images used (McGarigal & Marks, 1995). In higher resolution imagery where the smallest patches can be identified and distinguished, it is reasonable that Edge Density will be higher as more edges are identified and quantified. For many landscape ecological studies, the presumed importance of spatial pattern is related to edge effects. For instance, one of the most significant consequences of forest fragmentation is an increase in edge effects and adverse effects of this phenomenon on core sensitive species. With so much importance placed on edge effects in forest fragmentation studies because of the significance of edges on the species present in the area, it is important to note that, measurements of edge density are highly variable within a single landscape. Variation in edge density measurements depends on the spatial resolution of the imagery used for the analysis (Figure 5). Figure 5 shows that Edge Density values for Landsat and Sentinel are very different from those of NAIP and UAV images because of differences in spatial resolution. The increased capture of small forest patches by high resolution imagery, and therefore the increased ED values captured by NAIP and UVA images, demonstrates the value of high resolution imagery. However, given that differences between resolutions of NAIP and UAV imagery are not vast, the relatively large difference in their corresponding ED values, raises questions about effects of canopy shadow (Asner & Warner, 2003).

Fig. 5: Increasing Edge Density values with increasing spatial resolution. This means that there is a correlation between spatial resolution and edge density metric values

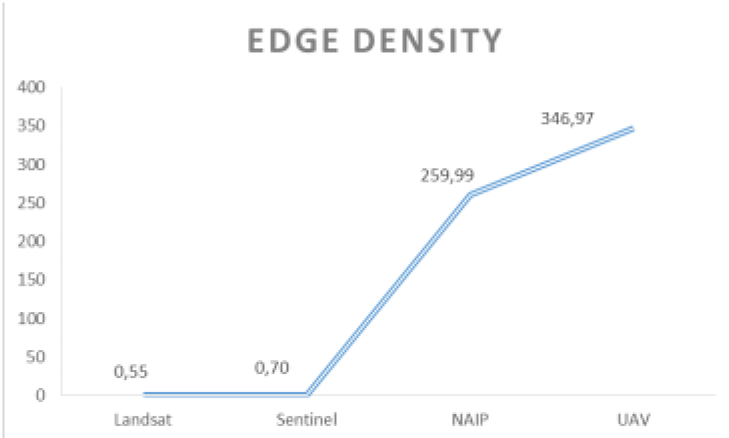


Table 4: Classification Accuracy Assessment. Created using randomly selected points in the forest (250 points) and non-forest (250 points) areas of the classified NAIP, UAV, Sentinel and Landsat images.

		REFERENCE		
CLASS	Forest	Non- Forest	Row Total	Overall Accuracy
		NAIP		
Forest	207	40	247	
Non-Forest	14	189	203	0.792
Column Total	221	229	500	
		UAV		
Forest	183	62	245	
Non-Forest	81	174	255	0.714
Column Total	264	236	500	
		SENTINEL		
Forest	179	79	258	
Non-Forest	74	168	242	0.694
Column Total	253	247	500	
		LANDSAT		
Forest	163	108	271	
Non-Forest	71	158	229	0.64
Column Total	234	266	500	

CLASSIFICATION

Canopy shadows, higher in forested regions, refer to reflectance of vegetation. With fine resolution imagery, canopy shadows become increasingly easily detected and can be erroneously categorized as part of the forest structure by computer algorithms such as the 'Maximum Likelihood' classification tool in ArcGIS, that was used for this analysis. This is why a closer look at the NAIP and UAV images show a decrease in classification accuracy of the UAV imagery (Table 4). Thus, it is important for ecologists and policy makers to be prudent in the use of high resolution imagery, especially with regard to edge density values from fragmentation analyses.

LSI measures the overall shape of the landscape with values close to 0 indicating that the landscape has a simple shape with higher aggregation. Values that are far from 0, like those of the NAIP and UVA (Table 1), show that the landscape has a complex shape with dispersed patches, and not as aggregated as indicated by the values of the Landsat and Sentinel images. This result is contrary to the findings of Aithal *et al*. (2012) suggesting that fragmentation metrics based on shape like the LSI, are not sensitive and behave similarly across all spatial resolutions. From our results, LSI varies significantly across resolutions and should therefore be interpreted cautiously. LSI is dependent on spatial resolution because more patches, within the same landscape, are identified with higher resolution imagery, exposing the dispersion within the landscape (Boyle *et al*., 2014).

The radius of gyration is a measure of the average distance an organism can move within a patch before encountering the patch boundary from a random starting point. It is therefore a measure of landscape connectivity important in conservation studies for assessing health of species populations. Results from Table 1 show that, the radius of gyration is very sensitive to spatial resolution. Between the two high-resolution images used (NAIP and UAV), the difference in this metric is not significant but can pose a problem if conclusions are drawn from images of very different spatial resolutions such as between a Landsat image and an UAV image. Given the subtle difference in values for the NAIP and UAV imagery but the stark difference between coarser resolution Landsat and Sentinel images, conservationists can use the relatively cheaper NAIP imagery in connectivity studies involving inference from this metric, compared to the more expensive UAV imagery. This is important for connectivity studies that determine the abundance of species within an area as the more readily available and cheaper NAIP imagery, gives similar results as the more expensive UAV imagery.

An important part of our results lies in the values of the *Cohesion* metric. A Cohesion value of 100 is an indication of clumpiness or connectivity of the landscape patches. Values close to 0 indicate highly unconnected fragmented landscapes. Our results show that the Cohesion values in the landscape were 78.11, 78.39, 79.05 and 78.64 for Landsat, Sentinel, NAIP and UAV images respectively. A trend cannot be identified in the measurement of these values with respect to spatial resolution. Given that there is no significant difference in the metric values, it can be concluded that Cohesion is not sensitive to spatial resolution. This result is consistent with the findings of Aithal *et al*. (2012) who found that Cohesion results were similar irrespective of spatial resolution and therefore concluded that Cohesion is independent of spatial resolution. Given this result, the use of Cohesion as a metric of aggregation is useful since the result is independent of the spatial resolution of the satellite imagery used. It should be noted, however, that other metrics indicated higher fragmentation in the landscape, especially the high-resolution metric values from NAIP and UAV images. Thus, an average Cohesion value of 78, an indication of aggregation, is not a good representation of the level of fragmentation in the area.

Conclusion

Our results highlight the fact that differences in satellite image resolution used in fragmentation analyses are not trivial, and can reliably assess significant differences in patch metrics. Consequently, these differences are likely to influence interpretations of fragmentation metrics, which can directly impact populations of species within an ecosystem. Our results have shown that it is critical for every researcher to tailor spatial imagery needs according to objectives of the research and that higher resolution images do not always guarantee higher accuracy and better interpretations. It is important that future research identify specific threshold resolutions, above which high image resolution ceases to be useful for those specific objectives.

Assessment of the classification accuracy of remote sensing images remains very important. It is important that remote sensing researchers not assume that high resolution images automatically imply high classification accuracies. Different studies have found different effects of spatial resolution on image classification. Perhaps, it is important for future studies to accurately identify effects of spatial resolution on classification accuracy, on the premise of the field of interest such as for either marine, coastal, or terrestrial studies. Differences in effects of spatial resolution on classification accuracy might be more apparent if specific study areas are characterized.

In our study, NAIP imagery proved to have a higher classification accuracy compared to the higher resolution UAV imagery. The classification of the area based on the NAIP imagery was a better representation of the state of the area, given that ground data had been verified. Whereas the UAV imagery misclassified certain non-forest areas, NAIP imagery more accurately classified forests and non-forest areas. This effect shows the tendency of very high resolution imagery to produce canopy shadows that lead to false classifications. It is important for conservationists to do ground studies in order to produce better training data for forest classifications of high resolution imagery.

Despite high costs of high-resolution imagery, our results show its significance in detection of smaller patches. For measurements of connectivity within landscapes where small patches serve as stepping stones for most species within the larger ecosystem, it is important for conservationists to consider the spatial resolution of images used in the analysis. Also, in studies where forest loss detection is a primary aim, it is important to consider the resolution of images used in fragmentation analysis as they influence results. Preferably, high resolution images should be used in such studies.

Our study, based on 30 m, 20 m, 10 m, 0.25 m and 0.03 m spatial resolution images, missed some important intermediary information. Considering the poor performance of the coarse resolution images (30 m and 20 m) and improvement of the accuracy of 0.25 m over 0.03 m resolution images in identifying forest patches, it will be expedient to know if an image of between 5 m and 10 m spatial resolution, can perform even better. The inclusion of an image with spatial resolution between 5 and 10 m, will be helpful in illuminating this research interest.

The study area for this research consists of agricultural, forest and low density residential areas. With agricultural and forest areas having similar reflectance spectra, but vastly different from the reflectance spectra of non-vegetation, our study area might be missing some important spectral information. Higher resolution imagery might be more convenient in areas with relatively denser residential constructions, compared to our study area. Moving forward, it is important for this research to be replicated in other areas with differing vegetation and non-vegetation combinations, to assess the accuracies of high and coarse resolution imagery.

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Appendix C

Understanding changes in Wood Thrush and Ovenbird populations in Virginia – the role of forest fragmentation and connectivity

Abstract

Yearly summaries of the American Breeding Bird Survey (BBS) indicate that populations of many North American bird species are in decline. Determining the causes of these declines is the focus of much current research in avian conservation. Forest fragmentation has been linked to declines in populations of many species. In this study, we explore effects of forest fragmentation and connectivity as well as slope and physiographic features on two migratory bird species- the Wood Thrush (*Hylocichla mustelina*) and the Ovenbird (*Seiurus aurocapilla*). We used the Markov Chain Monte Carlo Generalized Linear Mixed Models to identify factors in surrounding areas of different sizes that affected changes in Wood Thrush and Ovenbird populations between 2001 and 2011. Results indicate that forest fragmentation has a significant impact on population sizes of Wood Thrush and Ovenbirds in Virginia and that recent changes in Virginia's landscape have had negative impacts on the populations of these bird species. Specifically, slope characteristics that influence rates of urbanization are correlated with changes in bird populations. The Coastal Plain region in Virginia contributes significantly to the populations of both the Wood Thrush and Ovenbird currently.

1.0 Introduction

In order to accurately evaluate effects of forest fragmentation on species in a forest ecosystem, it is crucial to define forest fragmentation appropriately (Bogaert et al., 2011; Lafortezza et al., 2008). Since 1980, landscape ecology literature has been populated with varied definitions of forest fragmentation. For instance, whereas Wilcove et al. (1986) defined forest fragmentation as the process whereby a large, continuous area of habitat is reduced in area and divided into two or more fragments, Wiens (1989) defined it as an alteration of the spatial configuration of habitats that involves external disturbance that alters the large patch so as to create isolated or tenuously connected patches of the original habitat. Lord and Norton (1990) focused on the fact that forest fragmentation disrupted continuity while Groom and Schumaker (1993) emphasized that forest fragment led to the conversion of natural vegetation to new land uses.

These varied definitions show both the causes and effects of forest fragmentation and indicate that whereas different processes could lead to forest fragmentation, a visible effect of the phenomenon is the creation of many smaller forest patches compared to a previously single and/or larger forest patch. The definition of forest fragmentation as a remnant of vegetation patches surrounded by a matrix of different vegetation and/or land use, by Saunders et al. (1991) brings a new perspective by shifting focus to effects of the changing characteristics of the surrounding vegetation, prompting inquiry into how the process of forest fragmentation affects biological species that either live in core forest areas or edge areas.

Forest fragmentation consists of three main components that lead to the decline of biological diversity: 1) edge effects, 2) reduction in habitat patch size, and 3) increasing isolation of habitat patches (Andren, 1994). Concerning biological diversity and conservation, Caughley (1994) highlights two non-overlapping scientific paradigms: small population paradigm and declining population paradigm. Of these two, this study focuses on the latter which seeks to identify the processes by which populations are driven to extinction by agents external to them. The declining population paradigm is rooted in empiricism and therefore focuses on detecting, diagnosing and halting population decline (Caughley, 1994). The tenet of this study is that there is a tangible cause for the decline in populations of biological species and that probing reveals this cause. Forest fragmentation is identified as a probable agent of species decline (Diamond, 1984) and therefore forms a key part of this study.

Although impacts of forest fragmentation may vary among different biological species, it is generally accepted that reduction in forest patch sizes and decreasing connectivity between forest patches, negatively affect populations of specialist bird species that prefer core forest areas (Nour et al., 1999). Robinson et al. (1995) also showed that there is a high correlation between forest cover and rates of forest fragmentation, with higher forest fragmentation occurring in areas that have experienced forest cover reductions. Factors such as rates of predation and parasitism, prevalent more in certain forest types and conditions, contribute to effects of forest fragmentation on bird species (George and Dobkin, 2002).

Forest fragmentation not only affects population densities of forest bird species, but also affects their reproduction and dispersal (Walters et al., 1999; Donovan and Flather, 2002). Forest fragmentation can impact bird species negatively by reducing food resources available to birds as a result of a reduction in core forest patch sizes. Reduction in food resources for birds also have a consequent negative impact on their fecundity, causing a decline in their numbers within a particular area (Zanette et al. 2000, Lampila et al., 2005). Forest breeding birds such as Wood Thrush and Ovenbird species, with particular preference for core forest areas, are most negatively affected by forest fragmentation.

The Wood Thrush (*Hylocichla mustelina*) breeds in North America, migrating to Central America during the winter (James et al. 1984). Recently, Wood Thrushes have become a species of conservation concern because of their narrow tolerance of certain microclimatic conditions and their preference for the interior of moist deciduous forests, a forest condition that is fast declining due to timber harvesting activities. Wood Thrushes are highly sensitive to forest fragmentation due to increased nest predation and brood parasitism by the Brown-headed Cowbird (Brittingham and Temple, 1983) in forest fragments.

Ovenbirds (*Seiurus aurocapilla*) are also fragmentation-sensitive Neotropical migrants (Morimoto et al., 2012) with a preference for mature deciduous or mixed forests with closed canopies and sparse understory (Van Horn and Donovan, 1994). Pairing success of male Ovenbirds is lower in forest edges and small forest patches, compared to areas with large core forest patches (Bayne and Hobson, 2001). Numbers of Ovenbirds are also dependent on the surrounding vegetation patterns as small forest patches with surrounding agricultural lands have fewer Ovenbirds compared with large contiguous forest patches (Porneluzi et al., 1993; Bayne and Hobson, 2001). Mechanisms of forest fragmentation effects on Ovenbirds are not completely understood as nest predation rates of Ovenbirds in large contiguous forests and in fragmented forests have been shown to be similar in certain studies (Bayne and Hobson, 2001; Rudnicky and Hunter, 1993; Andren, 1995).

There are specific threshold sizes for the patches formed by habitat fragmentation, beyond which particular species within the habitats cannot survive (Homan et al., 2004). The relationship between habitat fragmentation and population density of the Wood Thrush and Ovenbird within Virginia can be used to identify impacts of forest fragmentation on the species in the state. Given that human development is continuing, effective planning and management are needed to conserve biodiversity by determining minimum-area breeding requirements for Neotropical migrants such as these species, and to identify landscape and vegetation features that support viable breeding populations (Donnelly and Marzluff, 2004; Reale and Blair, 2005). There is an opportunity to compare populations of these bird species with changing forested land areas over the years to identify correlation and subsequent impacts on the species. The goal of this research is to determine the impact that changes in forested land areas in Virginia, have had on Wood Thrush and Ovenbird populations.

Largely due to the ease with which humans develop lowland areas compared to highland forests that require more intensive human labor and therefore higher costs, logging and agricultural conversions are more pervasive in lowland regions (Barlow et al. 2006; Schnell et al., 2013). Especially for terrestrial birds that thrive best in lowland forests, the trend of increased logging in these regions are a threat to their survival (Sam et al., 2014). Wood Thrush and Ovenbirds breed in mature and contiguous deciduous and mixed forests and are less successful in fragmented forests. Since logging is a significant cause of forest fragmentation (Schleuning et al., 2011), it is likely that populations of Wood Thrush and Ovenbirds will be lower in lowland forest areas.

The type of landscape (forest, rural, or urban) of an area can interact with reserve size, amount of regional forest cover, and nest-site availability to influence reproductive success and patterns of occurrence of birds. In urban regions where human populations are greatest, landscape scale studies of birds are very important (Feldman and Krannitz, 2004; Reale and Blair, 2005; Grimm et al., 2008; Forman, 2008). Irregularly patterned landscapes such as those formed by forests that grade into either woodlands or wooded residential lots are less susceptible to forest edge effects commonly associated with bird population declines, compared to forest patches embedded in other kinds of landscapes (Fischer et al., 2004; Donnelly and Marzluff, 2004; Morimoto et al., 2012).

Trimble (1964) showed that topography creates spatial variation within the landscape that influences forest productivity. Spatial variation in a landscape has bottom-up effects that affect food web systems in forest ecosystems (Seagle and Sturtevant, 2005). Virginia’s five physiographic regions (the Appalachian Mountains, Blue Ridge, Valley and Ridge, Piedmont and Coastal Plains) provide substantial differences in forest development and productivity resulting in their capacity to influence the populations of forest species, especially forest interior species that feed on the detrital forest floor.

Much of the research on effects of habitat fragmentation on Ovenbirds have been conducted in agricultural landscapes with relatively low regional forest cover, or in landscapes undergoing reforestation (Flaspohler et al., 2001; Mazerolle and Hobson, 2002; Vanderwel et al., 2007; Nol et al., 2005; Perot and Villard, 2009). It is the goal of this study to determine effects of different forest structures and physiographic characteristics on Ovenbird and Wood Thrush populations in the 60%-forested urbanizing region of Virginia.

Between 1978 and 1987, Breeding Bird Survey (BBS) data that collates information across the breeding range of the Wood Thrush and Ovenbird species in United States and Canada, showed a 4% decrease in bird populations. While Holmes and Sherry (1988) reported an annual decline of 3.6% among bird populations between 1969 to 1986, Wilcove (1988) identified small increases in the population numbers using similar data, with Serrao (1985) reporting very little or no change. It is therefore important to identify the trend of population numbers of Wood Thrush and Ovenbird species in recent years and what factors are influencing the trend.

In this study, the pattern of change in population of the Ovenbird and Wood Thrush species in Virginia between 2001 and 2011, is established. Relationships of trends to slope, forest type and physiographic region in Virginia are also examined at both local and regional levels. We categorize forests in Virginia as either Core, Connected or Fragmented, based on pixels surrounding every forest pixel in the landscape imagery and establish whether changes in forest types from 2001 and 2011, correlate with trends of change in species population.

We identify factors most likely to affect populations of Wood Thrush and Ovenbird as one moves further away from the survey routes where their numbers are measured. While the BBS is viewed as a scientific tool to aid in wildlife conservation and management decision making, it is subject to bias like any population survey (North American Breeding Bird Survey Panel, 2000). Although the target sampling frame for the BBS should include all habitats, for practical reasons, the BBS sampling frame includes only habitats near secondary roads. Because the BBS relies solely on secondary roadside counts, a key assumption made is that roads do not bias the measurement of relative abundance by attracting or repelling birds in the area and that off-road trends are the same as on-road trends (North American Breeding Bird Survey Panel, 2000). To validate the assumption that birds are not attracted or repelled by roads, it is important for decisions based on the BBS to incorporate information from and characteristics of areas surrounding survey routes. To accomplish this, we created buffers with differing radii around the BBS survey routes and identified which factors affect Ovenbird and Wood Thrush populations most significantly in each of the buffer zones. Results of this study are especially important for policies that are site-specific and decisions that aim to improve quality of habitats with regards to human-bird interactions.

2.0 Methods

2.1 Study area

Out of the more than 900 bird species that can be found in North America, Virginia provides natural habitat for about 400. Virginia serves as either a permanent or transitory home to approximately 45% of the bird species in America, out of which Virginia’s Wildlife Action Plan has identified about 96 species to have declining populations (Free, 2013).

Virginia, a constituent state of the United States, is bordered by the states of Maryland, Tennessee, Kentucky, North Carolina and West Virginia (Figure 1). Virginia’s land cover is approximately 60% forestland, including both pine and hardwood forests. Mountainous areas, especially in the Appalachian region, contain tracts of various coniferous species and hardwoods such as hickory and oak. Virginia’s very high avian diversity is due to varied habitats found in its varied physiographic regions - Coastal

Plain, Piedmont, Appalachian Plateau, Blue Ridge and Valley and Ridge (Nott et al., 2008). Virginia's oak-hickory and oak-pine communities offer favorable habitat for both Ovenbirds and Wood Thrush.

2.2. Breeding Bird Survey

Initiated in 1966, the North American Breeding Bird Survey (BBS) is an international monitoring program, jointly coordinated by the USGS Patuxent Wildlife Research Center and the Canadian Wildlife Service, to track the status and trends of North American bird populations. In June every year at the height of the avian breeding season in the U.S. and Canada, skilled avian identification experts collect data on avian populations along specified roadside survey routes. Approximately 24.5 miles long (Figure 2), each roadside survey location has several stops situated 0.5-miles apart. At each stop, bird populations are counted and recorded over a 3-minute period. In Virginia, there are about 88 survey routes, a few of which have become inactive in recent years (Figure 2). The numbers of birds detected at each survey route in 2001, 2006 and 2011, were used in this analysis.

2.3. Methodology

The study is designed to identify factors driving decline in Wood Thrush and Ovenbird populations in Virginia between 2001 and 2011. Specifically, the effects of forest fragmentation, connectivity, slope and physiographic region on the species' populations are determined. The total count for Ovenbirds and Wood Thrushes at each of the 77 survey routes across Virginia at three time points (2001, 2006 and 2011), Virginia's 5 physiographic regions and 8 slope categories are investigated. The effects of forest fragmentation and connectivity are determined by calculating and comparing the changes in amounts of core, connected and fragmented forests. Percentages of physiographic regions and slope at different scales are also examined as potential factors driving change in the abundance of Wood Thrush and Ovenbirds.

Five buffers of different radii were created around each survey route; 25 miles, 20 miles, 15 miles, 5 miles and 1 mile (Figure 3). Three National Land Cover Dataset satellite images with a 30 by 30 m resolution were used, each representing 2001, 2006 and 2011. We classified forest patches in Virginia as either ‘Core’, ‘Connected’ or ‘Fragmented’ depending on neighboring pixels surrounding forest pixels (Figure 4). ‘Core’ forests include pixels that are completely surrounded by other forest pixels while forest pixels surrounded by more than 60% but less than 100% forest pixels are labelled ‘Connected’. ‘Fragmented’ forest areas include forest pixels surrounded by less than 60% other forest pixels. We then calculated the percentages of ‘Core’, ‘Connected’ and ‘Fragmented’ forests in each of the five different buffer zones created around each survey route. This process was applied three times to account for the changes in forest fragmentation between 2001 and 2006 and then between 2006 and 2011.

We categorized Virginia’s topographic slopes into 8 groups (Table 1), with 1 being the most gentle slope (between 0 and 2.5%) and 8 being very steep (above 60%). Percentages of the differing slope categories in each buffer zone around a survey route, were calculated in ArcGIS using the Zonal Statistics tool. Virginia’s five physiographic regions (Appalachian Plateaus, Blue Ridge, Valley and Ridge, Piedmont and Coastal Plain) were overlaid as a vector dataset (Figure 5) on each of the 5 maps of Virginia representing the 5 different buffer zones used in the study. In each of the 5 different buffer zones, we calculated the percentages of the five physiographic regions to identify how much of the physiographic region was in the buffer zone. Slope and physiographic regions were examined because of their potential to affect forest fragmentation and also influence Wood Thrush and Ovenbird populations.

The population abundance of the Wood Thrush and Ovenbird, as recorded by the BBS, in 2001, 2006 and 2011 along the survey routes were noted to identify the general trend of the species’ population. Using the Repeated Measures ANOVA, we sought to understand influences on populations of the Ovenbird and Wood Thrush in Virginia between 2001 and 2011. Repeated Measures ANOVA is an extension of the dependent t-test and is equivalent to the one-way ANOVA test, specific to related but independent factors. The Repeated Measures ANOVA was conducted using the R software under the hypothesis that the population means of the wood thrush and ovenbird do not change between 2001 and 2011.

$$H_0: \mu_{2001} = \mu_{2006} = \mu_{2011}$$

$$H_{A:} \text{ at least two means are significantly different}$$

In repeated measures ANOVA, the independent variable has categories called levels or related groups (Keselman et al., 1999). In this design, the independent variable is time and the different levels are the three time points that data on population numbers of Wood Thrush and Ovenbirds are collected; 2001, 2006 and 2011. The treatment variables are three different forest types – Core, Connected and Fragmented; 8 slope categories and 5 physiographic regions. The logic behind a Repeated Measures ANOVA is very similar to that of a between-subjects ANOVA but a repeated measures ANOVA has the advantage of reducing error variability by further partitioning the error term, thus reducing its size and consequently, increasing the power of the test to detect significant differences between means (Keselman et al., 1999).

In the R software, the Markov Chain Monte Carlo Generalized Linear Mixed Models (MCMCglmm) package with a Poisson distribution was used. An MCMCglmm was used because it provides a more flexible way of inputting a covariance structure for Repeated Measure ANOVA with a count response variable. A Bayesian data analysis model with a 95% probability interval was employed, since the data used are subject to random variation. Prior information employed includes the exemption of the core forest type, the Appalachian physiographic region and one slope category, from the model. The MCMCglmm model cannot include all the treatments of every variable if the treatment variables add up to 100. Since all the treatment variables were calculated as percentages, they all add up to 100 and thus, one treatment of each variable had to be excluded from the model. Each treatment variable chosen for exclusion is based on a model selection from different combinations. Specifically, a deviance information criterion (DIC) is calculated from MCMCglmm with a lower DIC number identifying a better model.

For the 3 forest types, only connected and fragmented forest types were included in the model because changes between 2001 and 2011 for these two were significant. These two forest types make up the research focus on forest fragmentation and connectivity. For the physiographic regions, a comparison of the combinations of ABCD, ABCE, ACDE, ABDE, and BCDE is made with Appalachian Plateau, Blue Ridge, Coastal Plain, Piedmont and Valley and Ridge represented by the letters A,B,C,D and E respectively. For the eight slope categories, the combinations of 1234567, 1234568, 1234578, 1234678, 1235678, 1245678, 1345678, and 2345678 are compared with each figure representing a slope category. The three year time periods and the 2 forest types are included in the model and run separately for Wood Thrush and Ovenbird multiple times, to identify which model has the lowest DIC. For the Ovenbird model, the model excluding the Appalachian Plateau and Slope category 3 had the lowest DIC and was therefore chosen. The model excluding the Appalachian Plateaus and Slope Category 2 was chosen for the Wood Thrush model.

With the number of iterations, burnin and thinning set at 80,000, 10,000 and 10, respectively, the model is run with traceplots drawn for both fixed and random effects in the model. Traceplots are essential for assessing convergence and diagnosing chain problems. Normal traceplots should have no trend and the values should jump randomly above and below the central value. A posterior density plot, which is an estimate of the posterior distribution based on the 70,000 samples used, has to be examined for every variable used in the model. Bayesian analyses do not provide single outcome values but an interval with a probability that this interval contains the regression coefficient.

3.0 Results

Between 2001 and 2011, populations of both Wood Thrush and Ovenbirds species have declined. On the average, the lowest numbers of Wood Thrush and Ovenbird species detected in each survey route have not gone any lower than previous years. However, the highest numbers detected in each survey route have varied considerably between 2001 and 2011 (Figure 6). The average number of Ovenbirds recorded per survey route in 2001 was approximately 13, compared to 12 in 2011. There is a more a drastic change among Wood Thrushes as the average number of Wood Thrushes recorded per route in 2001 was 13 but declined to 7 by 2011.

Table 2 shows the significance of changes in forest fragmentation in Virginia between 2001 and 2011 in the various buffers. Whereas, changes in connected and fragmented forests are significant in the 15, 20 and 25 mile buffer radius, only connected forests are significant in the 5 mile buffer radius. In the 1 mile buffer radius, none of the forests had significant changes. Changes in core forests are not significant between 2001 and 2011. Generally, the trends are similar at all buffer sizes, with core forest declining and both connected and fragmented forest increasing (Figures 7, 8, 9, 10 and 11), although these trends were statistically significant only for the latter two forest types at larger scale buffer sizes.

There was no visible trend in changes of percentages of the physiographic regions as buffer radii differed. Generally, percentages calculated for physiographic regions for each buffer radius around survey routes increased as the buffer radius increased. Percentages of the areas around each survey route also increased as buffer radius increased but there was no recognizable trend between survey routes or from year to year.

Using the 1 mile buffer radius, Ovenbird numbers were significantly lower in fragmented forest compared to other forest types, and significantly higher in the Coastal Plain physiographic region than in all the other regions (Table 3). Also, slope categories 2, 5 and 8 have negative relationships with Ovenbird numbers, whereas 1, 3, 4, 6 and 7 have positive relationships. Only the negative relationship with slope category 5 (9 -15%) was significant. Ovenbird numbers did not differ significantly between time periods (Table 3). Wood Thrush abundance was significantly lower in 2011 than in 2001 (Table 4).

Within the 1 mile buffer radius, the only environmental variable that had a sizeable effect on Wood Thrush numbers was the slope category 4 variable (Table 4).

Results using the 5 mile buffer were identical to those for the 1 mile buffer. For Ovenbirds, numbers were lower where there was more fragmented forest, higher in the Coastal Plain compared to other regions, and reduced in slope category 5 (Table 5) . For the Wood Thrush, no variables other than year 2011 had a significant relationship to abundance (Table 6).

The Coastal Plain region had significantly higher numbers of Ovenbirds using the 15 mile buffer (Table 7), and again, Wood Thrush numbers were lower in year 2011 (Table 8). None of the variables analyzed had a significant relationship to Ovenbird numbers using the 20 mile buffer, and for Wood Thrush, the only significant variable again was the year, 2011 having significantly lower numbers than 2001 (Tables 9 and 10). Using a 25 mile buffer radius around the survey routes, Wood Thrush numbers were significantly lower in year 2011 compared to 2001 (Table 12).

4.0 Discussion

Populations of Ovenbirds and especially Wood Thrushes, can experience reductions as a result of nest predation and brood parasitism (Philips et al., 2005). Being habitat specialists preferring core forest areas, both Ovenbirds and Wood Thrushes are susceptible to effects of urbanization that leads to forest habitat fragmentation. Germaine et al. (1998) and Savard et al. (2000) showed that urbanization that results in forest fragmentation could significantly reduce densities of bird communities and most especially, specialist species like the Ovenbird and Wood thrush. While forest fragmentation leads to reductions in the population density of specialist species, the population density and diversity of more generalist species including brood parasites such as the Brown-headed Cowbird, simultaneously, increase (Kluza et al., 2000; Mancke and Gav, 2000).

The Breeding Bird Survey suggests that the population of the Brown-headed Cowbird, a brood parasite known to have adverse impacts on nesting of both the Wood Thrush and Ovenbird, increased from 2001 to 2011 in Virginia. The general increase in numbers of the Brown-headed Cowbird could be a contributing factor to the reduction in population densities of Wood Thrushes between 2001 and 2011.

Many studies such as Donovan and Flather (2002), Bohning-Gaese et al. (1993) and Haila (2002) hypothesized that population declines in migratory specialist species such as the Ovenbird and Wood Thrush, were caused by reduction and isolation of their breeding habitats. The three forest classes in this analysis, Core, Connected and Fragmented were categorized based on surrounding pixels and the connectivity of those pixels and as such, capture the forest conditions important to the Wood Thrush and Ovenbird.

As our results show a significant increase in fragmented forests across all survey routes from 2001 to 2011, the decline in populations of Wood Thrush in Virginia over that period parallels an increase in forest fragmentation. Forest fragmentation leads to the creation of small forest patch sizes which limit the food resources available for the bird species. Thus, the link between the decline in Wood Thrush populations and forest fragmentation in Virginia is consistent with the findings of Weinberg and Roth (1998) who found a direct relationship between forest patch sizes, reproductive success, and return rates for Wood Thrushes in Delaware.

Our results indicate a modest decline in Ovenbird populations between 2006 and 2011, whereas Wood Thrush declined significantly and consistently between 2001 and 2011. Since forest fragmentation significantly affected Ovenbirds in the 1 and 5 mile buffers, our results will be consistent with the results of Bayne and Hobson's (2001) study suggesting that pairing success of Ovenbirds can be influenced by habitat fragmentation. Ovenbirds avoid forest fragments which might be the reason for their modest decline compared to the significant decline in Wood Thrush populations.

Core forest areas are important for buffering Wood Thrush and Ovenbird species from the incursions of predators as a result of the shield provided by the high and closed canopy structure with small canopy gaps (Hagenbuch, 2010). Declining core forest areas can adversely affect interior specialists like the Ovenbird and Wood Thrush who primarily breed in large forest interior patches (Hostetler, 2016). However, our results show that the decline in core forest in Virginia is not significant. This means that the decline in Wood Thrush is caused by other factors such as increased forest fragmentation which leads to increased predation.

For many migrating interior forest specialists like the Ovenbird and Wood Thrush, although small forest fragments are not appropriate for breeding, they serve as stopover habitats and corridors that link core forest patches together (Hostleter, 2016). Our results show that the change in connected forests in all the buffer regions is significant which is important for ensuring the persistence of healthy Wood Thrush and Ovenbird populations.

Forest fragmentation does not always directly cause declines in populations of migrant species such as Ovenbird and Wood Thrush. Other nest predators such as raccoons (*Procyon lotor*) and the Eastern Gray squirrels (*Sciurus carolinensis*) are impacted by forest fragmentation. Philips et al (2005) purported that developed forest fragments increase availability of food and shelter for nest predators like raccoons (*Procyon lotor*) and squirrels (*Sciurus carolinensis*) depending on the topographic conditions of the area. According to Crooks and Soule (1999) and Odell and Knight (2001), domestic pets which either prey or disturb nesting birds are more abundant close to houses. Construction of residential homes has a significant correlation with topographic conditions in the area.

Reductions in populations of migrant habitat specialists such as the Ovenbird and Wood Thrush could be a result of increasing urban development which directly leads to increased rates of nest predation in forest fragments close to development sites, changes in habitat structure and composition, and consequently, avoidance of developed areas by birds due to human habitation (Kluza et al. 2000, Mancke and Gavin, 2000). Among effects of urban development, are increases in numbers of domestic pets from nearby homes in the area, which are likely to disturb nesting birds (Philips et al., 2005; Crooks and Soule, 1999; Odell and Knight, 2001).

Urban development is influenced by low costs of sloped land, and also by the risks, including losses of human lives and property, accompanying construction of cities in steeply sloped areas (Kechebour, 2015). Thus, while gently sloped lands are preferred for urban development, some steep terrain might be developed due to relatively lower costs, especially if risks to human life and property are not too high.

Landscapes change over time due to both natural and human-induced causes (Veech et al., 2017). It is therefore important that the Breeding Bird Survey (BBS) which is the preeminent source of data in North America for monitoring bird populations, employ survey routes that are representative of land cover composition and characteristics. Depending on habitat structure and surrounding avi-fauna within a landscape, specific bird species will either be present or absent, abundant or not (Veech et al., 2017). Thus, population trends of bird species are significantly dependent on habitat characteristics of survey routes. Given that the landscape of Virginia is constantly changing, it is important to consider characteristics of the habitat surrounding the BBS survey routes and identify which habitat characteristics best influence the populations of the birds at any given distance away from the survey routes. Hence, the 1, 5, 15, 20 and 25 mile buffer radii were used in this study to identify at what scales habitat characteristics are most likely to influence Ovenbird and Wood Thrush populations at different scales. Our results show that at local scales (1, 5 and 15 mile buffer radius), fragmentation, slope and physiographic regions have significant effects on Wood Thrush and Ovenbird populations. At the regional scale (20 and 25 mile buffer radius), forest fragmentation has no effect either due to the fact that forests no longer dominate the matrix or the species do not respond to the landscape at these large scales. Results of this study will provide policy makers with the needed information to enact measures both near and distant from survey routes, to ensure healthy populations of both the Wood Thrush and Ovenbird. For instance, because forest fragmentation is significant to Ovenbirds only at local scales, management efforts focused on improving conditions in local forest ecosystems rather than national or state forest policies, will be more effective in improving the abundance of Ovenbirds.

There was a similar trend in changes in core, connected and fragmented forest in every buffer area considered (Figures 7, 8, 9, 10 and 11). Core forest areas declined from 2001 to 2011 while fragmented forest areas increased, demonstrating a general increase in forest fragmentation across Virginia. This means that a singular policy aimed at decreasing forest fragmentation can be effective across the entire state of Virginia. For instance, Albers (1996) showed that economic research can rely on a model of net benefits that reflects or emphasizes the fragmentation or pattern of the landscape, rather than individual species or specific locations within the landscape. However, given that effects of all variables considered in each buffer zone on the populations of wood thrush and ovenbird differ, it is important that more specific deliberations are made, with particular species in mind. For example in the 5 mile buffer, fragmented forests, the Coastal Plain region and slope category 5 have significant effects on Ovenbird populations but none of these factors have significant effects on Wood thrush populations in the same buffer. Hence, results are dependent on the species under focus and policy makers should be cautious in the application of generalized concepts to all species.

4.1. Specific effects on Ovenbird Populations in Virginia

At local scales (i.e. 1 and 5 mile buffers), ovenbirds are less common in fragmented areas, more common in the Coastal Plain region and less common in slope category 5. BBS survey routes are roadside surveys which means that the count is influenced by conditions of forest edge areas. With a key result of forest fragmentation being an increase in edge areas (Sisk and Battin, 2002), fragmented forest types, usually characterized by small patch sizes and linear or irregularly shaped patches, are most likely present near survey routes where data on the birds are collected as they are more abundant near roads and pathways (Sisk and Battin, 2002). Therefore, roads likely create more negative impacts on forest interior species such as the Ovenbird, in the 1 and 5 mile buffer zones. The Ovenbirds' tendency to nesting only within forest interiors and rarely near forest edges is due primarily to lower habitat quality in forest edges (Ortega and Capen, 1999). At forest edges, vegetation structure limits availability of cover, nesting and foraging sites for Ovenbirds (Ortega and Capen, 1999).

The Coastal Plain, a low-relief terraced region that stair-steps down gently towards the coast and covers about 21% of Virginia, has higher densities of Ovenbirds than the other regions in the 1, 5 and 15 mile buffer zones. The Coastal Plain covers about 21% of Virginia and can be divided into inner and outer sections based on topographic features with the outer Coastal Plain characterized by an undulating to a nearly flat landscape (Fleming, 2012). A line through the town of Suffolk north to Gloucester Courthouse and Westmoreland County on the Potomac roughly divides the inner and outer Coastal Plain. It is therefore important for future research on the effect of the Coastal Plain region on Ovenbird populations in Virginia to take into consideration the differences in characteristics of the inner and outer regions.

During the Civil War, most of the old forests in the Coastal Plain were destroyed for lumber construction and firewood for the armies (Boutin, C., 2015). With about 46% of the Coastal Plain forested, urban development in large cities like Roanoke, Newport News and Norfolk has led to broad-scale forest fragmentation (Fleming, 2012). Currently, only 13% of the Coastal Plain region consists of unfragmented natural lands with high ecological integrity, with these areas limited to only the Great Dismal Swamp and the Fort A. P. Hill Military Reservation (Weber, 2007). This suggests that higher numbers of Ovenbirds occur in the Coastal Plain region despite high levels of fragmentation. However, there are natural mixed communities of the longleaf pine (*Pinus palustris*) and pond pine (*Pinus serotine*) woodlands that are largely restricted to the southern part of the Coastal Plain (Fleming, 2012). With Ovenbirds having a preference for mixed-pine hardwood ecosystems (Masters, 2007), it is typical for this region to have a significant positive effect on ovenbird populations.

Slope category 5 representing areas with slope percent greater than 9 but less than 15, has a significant negative relationship to Ovenbird numbers in the 1 and 5 mile buffer zones. The Southern Tier Central Regional Planning and Development Board (2012) defines a moderate slope to be between 10 and 15%, with anything below and above this range considered as gentle and steep respectively. For aesthetic and site drainage purposes, most residential and road constructions are carried out in areas with grades between 10 and 15% slope. Because steep slopes cause severe erosion, areas with large topographic gradients require re-sloping to create flatter land areas, which implies higher construction costs. Hence, areas in Virginia that are most likely to be developed, are areas that have slopes between 10 and 15%. Thus the relationship to slope category 5 may be due to greater residential development in such areas. As one moves further from the survey routes, in the 20 and 25 m buffer zones, there is increased possibility of finding areas with steeper slopes and therefore, the Slope 5 variable becomes less significant. Our results show that Ovenbirds are more responsive at the local scale compared to the regional scale.

In the 25 m buffer analysis, none of the variables analyzed had a significant effect on Ovenbird populations. There is only a moderate decline in Ovenbird populations between 2001 and 2011 despite forest fragmentation trends. Donovan and Flather (2002) argued that despite low fecundity success of Ovenbird populations in small forest fragments, a system of interacting subpopulations allows for local sink populations to persist as a result of immigration from other source populations. As such, the overall dynamics of source-sink systems are not necessarily impacted by poor reproductive success in forest fragments at larger scales especially if the birds avoid those small fragments (Donovan and Flather, 2002).

4.2. Significant effects on Wood thrush populations in Virginia

In the smallest buffer analyzed in this study, the 1 m buffer, Slope category 4 had a sizeable, albeit insignificant relationship to Wood Thrush abundance. Slope 4 represent areas with slope percent between 5 and 9%. According to the Agricultural Land Classification (2008), prime agricultural lands are characterized by slopes between 0 and 5% because they have low erosion hazards and are not prone to risks of damaging overflow. This means that forested lands with slopes in this category are more prone to agricultural conversion. Through the process of convective pre-heating, wildfires generally advance as slopes become steeper (Estes et al., 2017). As a result, it is highly likely that medium slopes, between 5 and 9%, present ideal conditions for mature forests in Virginia which might be why this slope category has a positive relationship to Wood Thrush numbers in Virginia in areas close to survey routes.

The year 2011 has a significant negative relationship to Wood Thrush abundance at all buffer sizes analyzed, and the year 2006 also has a negative relationship to Wood Thrush in the 20 and 25m buffer zones. This is best interpreted as indicating a decline in Wood Thrush populations in Virginia over the study period from 2001 to 2011 rather than indicative of annual variation. Thus populations declined while forest fragmentation increased. That Wood Thrush are not less common in fragmented forest than core forest suggests they do not avoid forest fragments, rendering their populations susceptible to adverse effects of fragmentation on their reproduction and survival. If the year effects instead represent annual variation, one possible cause of their variation is drought, which reduces availability of food resources for birds, making them more susceptible to predation and starvation (Hockman, 2017). According to the US Drought Monitor for Virginia, the state experienced severe to exceptionally severe droughts between 2007 and 2009. While drought severity reduced to abnormally dry conditions after 2009, Virginia again experienced severe drought in the latter months of 2010. These drought conditions might be responsible for a decline of Wood Thrush numbers being evident in 2006 and 2011.

Only in the 15 m buffer did the Coastal Plain region have a sizeable positive relationship to Wood Thrush abundance. The large composition of mixed-pine woodlands in the Coastal Plain region of Virginia can be a contributing factor to the positive significant effect of this region on Wood Thrush populations. Lang et al. (2002) found a positive correlation between mixed forests with high pine concentration and Wood Thrush populations. Hunter et al. (2001) reported that the South Atlantic Coastal Plain has one of the largest forested floodplains and the largest remnants of former longleaf pine and woodland ecosystems.

5.0. Conclusion

Our study shows that populations of Wood Thrush and Ovenbird in Virginia decreased between 2001 and 2011, forest fragmentation increased over this period, and that Ovenbirds are less common in forest fragments than in core forests. Increased fragmentation could account for the significant decline in Wood Thrush since our results show that they do not avoid fragments whereas Ovenbirds avoid forest fragments and have therefore only declined moderately. Our study also demonstrates how habitat conditions affect populations of Wood Thrush and Ovenbird. For example, slope has a significant effect on Wood Thrush and Ovenbird populations because it determines the intensity of development in an area. The Coastal Plain region harbors larger numbers of Wood Thrush and Ovenbird than other physiographic regions in Virginia as its mixed forested lands with high pine concentrations provide an ideal habitat for both the Wood Thrush and Ovenbird.

Future studies focusing on cowbird populations, a key brood parasite of Ovenbirds and Wood Thrush, can provide insight into decreasing population trends and their relationship to forest fragmentation. Other studies that add more variables, such as urbanization rate, drought occurrences and human population growth rate in Virginia to the model, will be beneficial in accurately determining factors affecting Wood Thrush and Ovenbird populations.

Our study identifies factors whose consideration may allow policy makers to make more effective decisions to increase Ovenbird and Wood Thrush populations in Virginia. In particular, use of differing buffer zones showed that these species respond to their environment at relatively small (1 -5 miles) spatial scales. The traceplots of our results show effectiveness of our model in identifying the effects of multiple variables over a time gradient. Thus, our model can be replicated and improved by other studies to ensure that effective decisions are made to improve bird populations.

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Figures

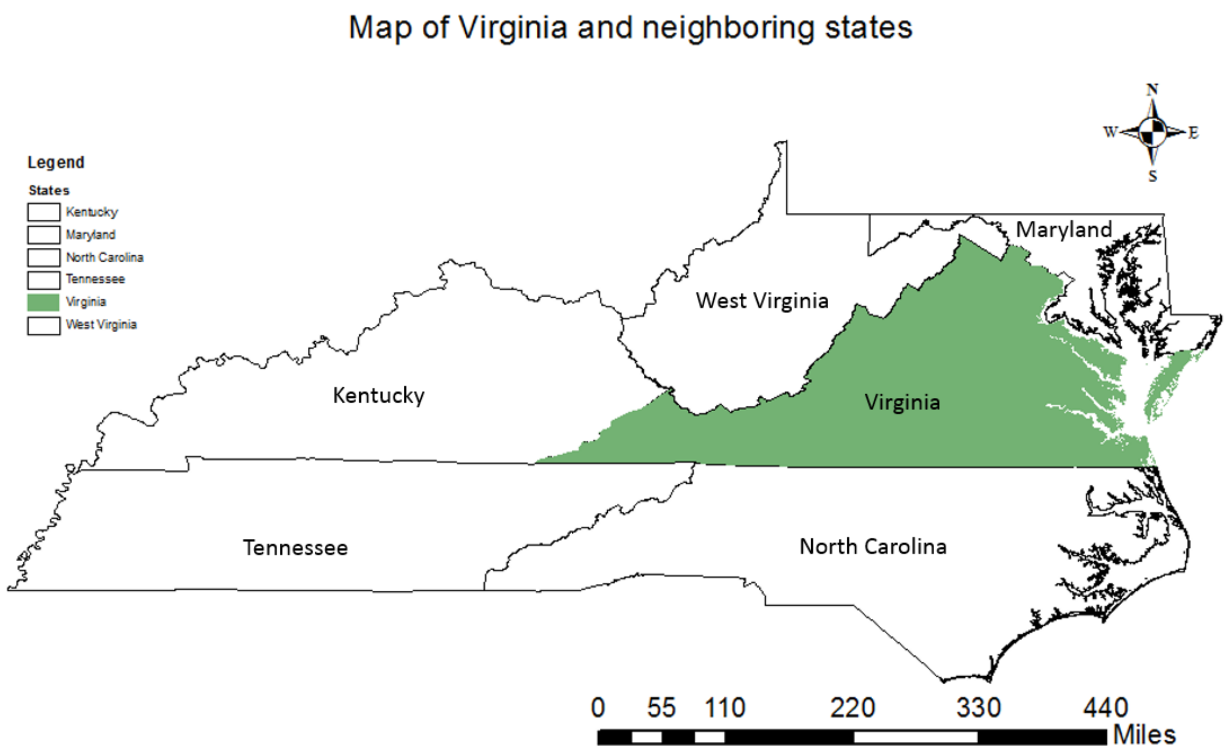


Figure 1. Map showing the state of Virginia and its neighboring states.

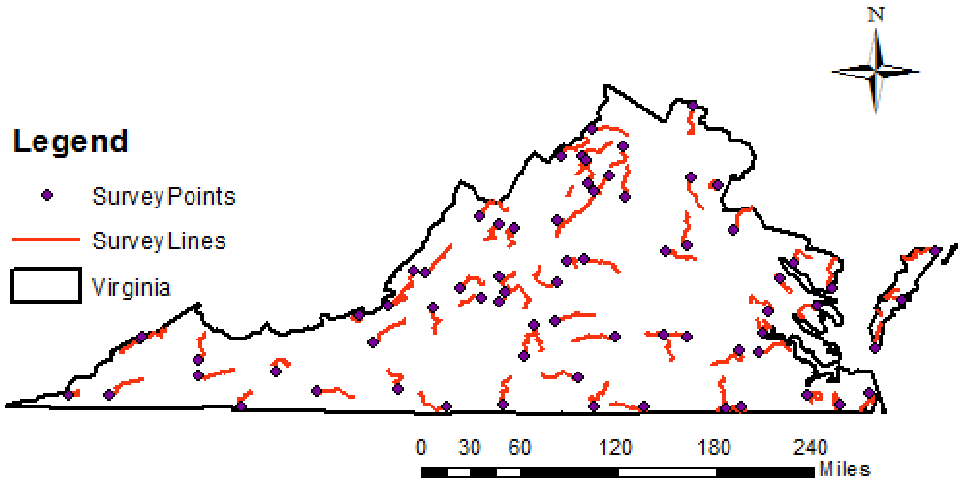


Figure 2. Map of Virginia showing the starting point of each survey route. Each survey route is about 25 miles long. Birds are counted every 0.5 miles along the route.

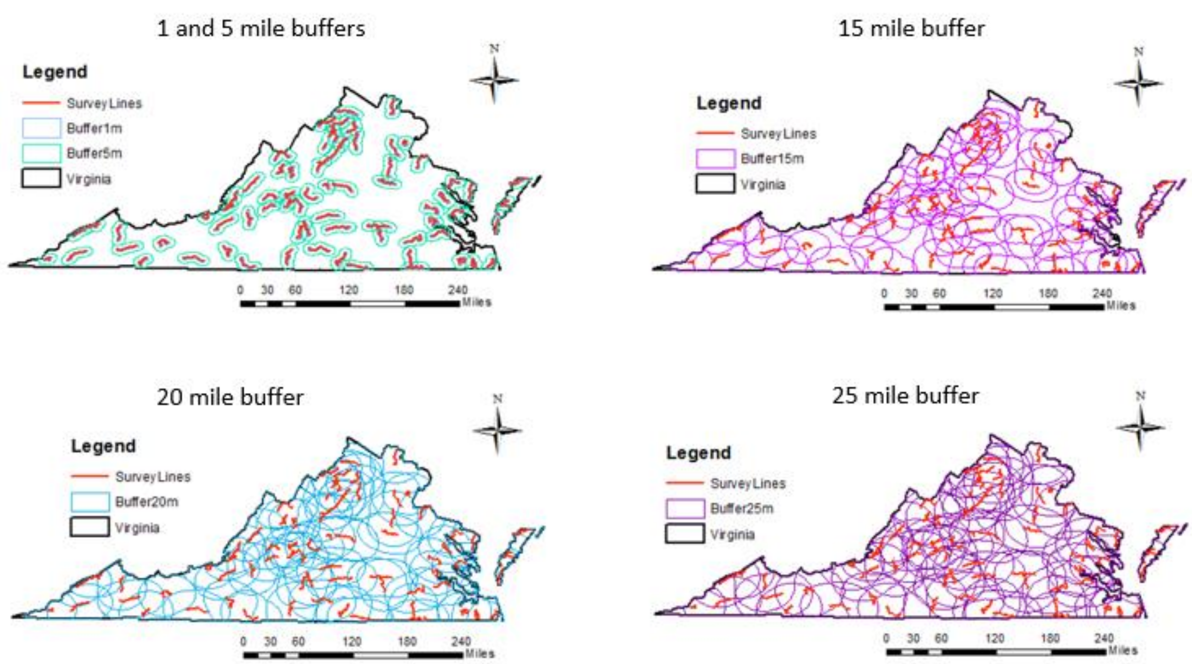


Figure 3. Map of Virginia showing survey routes and buffer zones created around them. While the 1 and 5 mile buffers have very little intersections, the 15, 20 and 25 mile buffers are more complicatedly intersected.

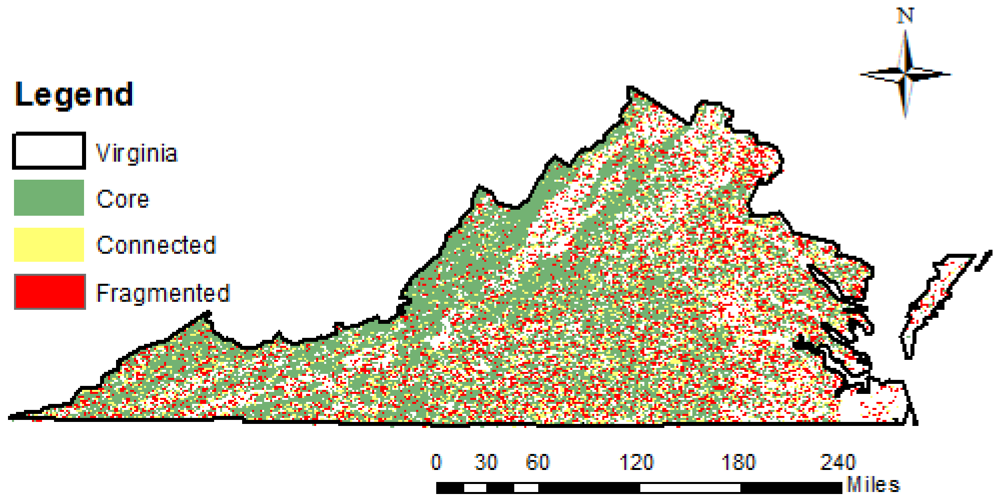


Figure 4. Map of Virginia in 2001 showing 3 forest classes; Core, Connected and Fragmented. Two other maps (not shown) showing the same forest classes were made for 2006 and 2011 to show forest fragmentation and connectivity.

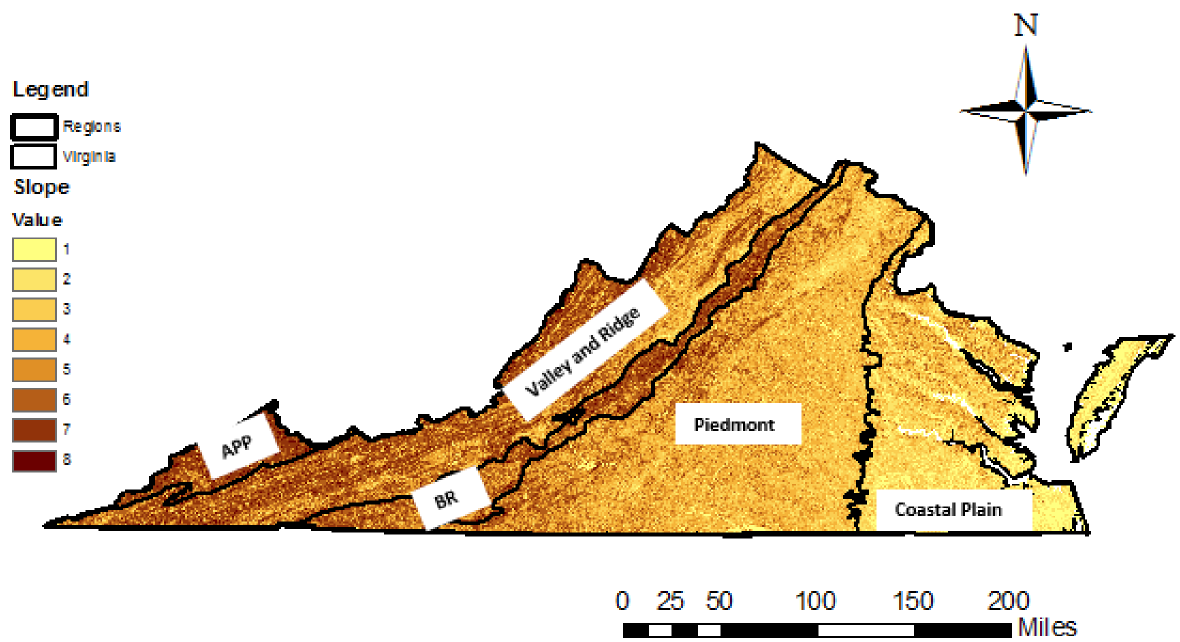


Figure 5. Map of Virginia showing the five physiographic regions (Appalachian Plateaus = APP, Blue Ridge = BR, Valley and Ridge, Piedmont and Coastal Plain) and 8 slope categories. Slope group 1 is very gentle and becomes steeper from 1 to 8 with 8 being the steepest.

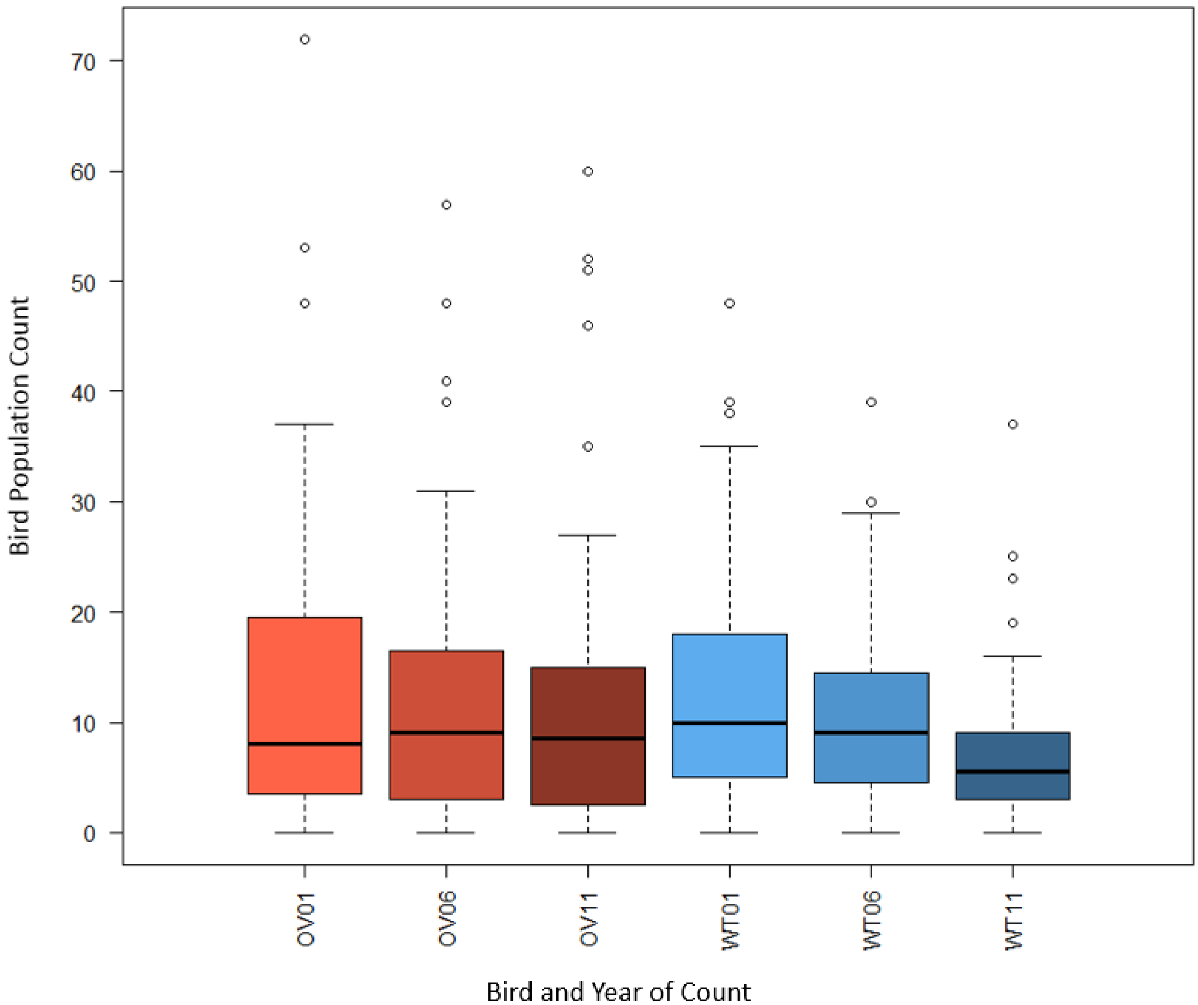


Figure 6. Populations of Ovenbirds and Wood Thrushes between 2001 and 2011. The red shades represent Ovenbird populations while the blue shades represent Wood Thrush populations. Between 2001 and 2011, the average number of individuals per survey route of Wood Thrush have declined.

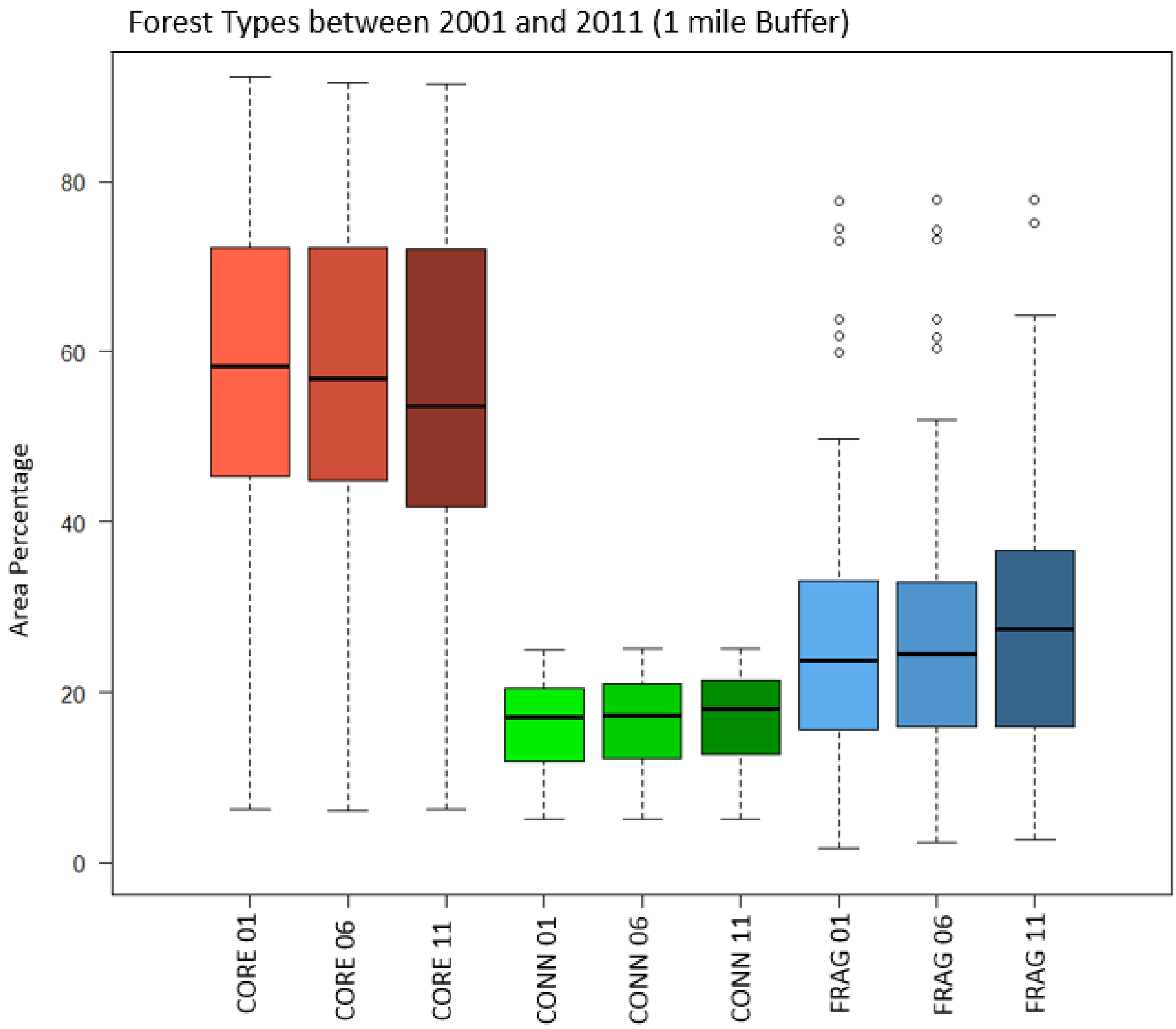


Figure 7. Changes in forest fragmentation in Virginia between 2001 and 2011 using a 1 mile buffer radius. While Core forest (red shades) generally declined between 2001 and 2011, both Connected (green shades) and Fragmented (blue shades) forest increased.

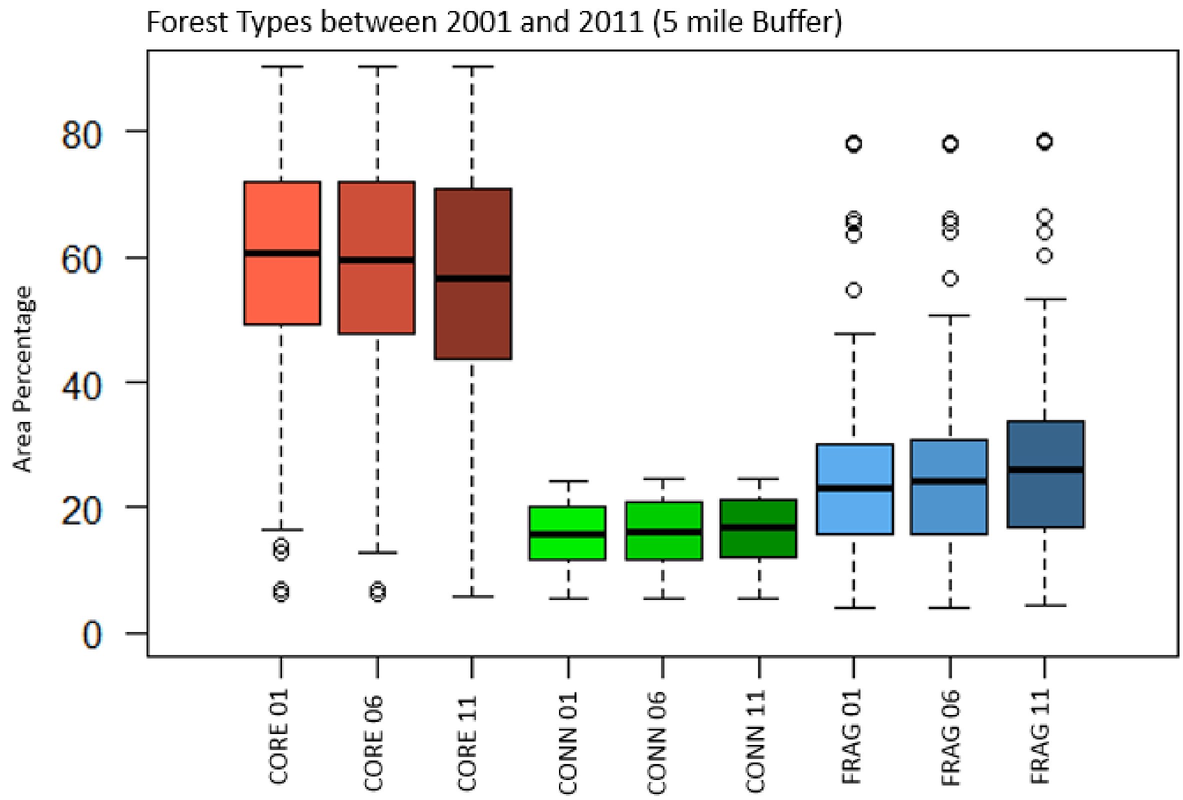


Figure 8. Changes in forest fragmentation in Virginia between 2001 and 2011 using a 5 mile buffer radius. While Core forest (red shades) declined between 2001 and 2011, both Connected (green shades) and Fragmented (blue shades) forest increased.

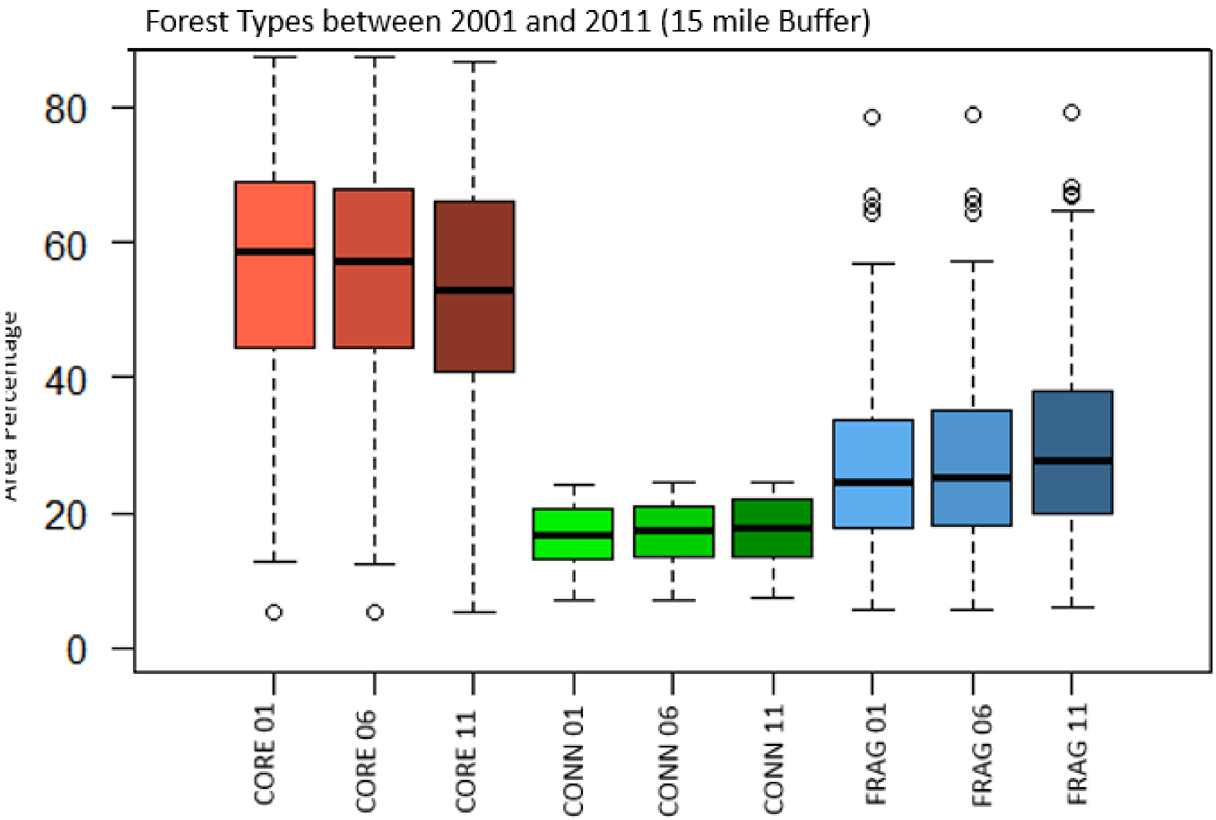


Figure 9. Changes in forest fragmentation in Virginia between 2001 and 2011 using a 15 mile buffer radius. While Core forest (red shades) declined between 2001 and 2011, both Connected (green shades) and Fragmented (blue shades) forest increased.

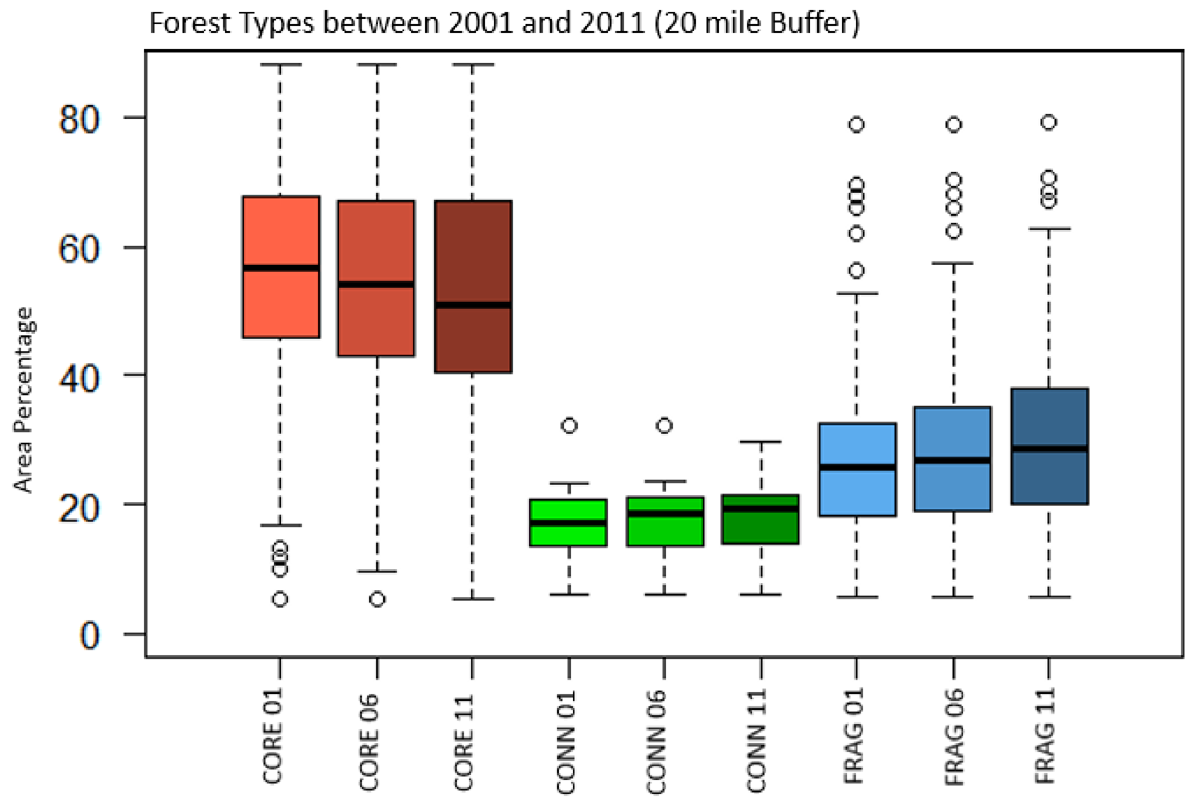


Figure 10. Changes in forest fragmentation in Virginia between 2001 and 2011 using a 20 mile buffer radius. While Core forest (red shades) declined between 2001 and 2011, both Connected (green shades) and Fragmented (blue shades) forest increased.

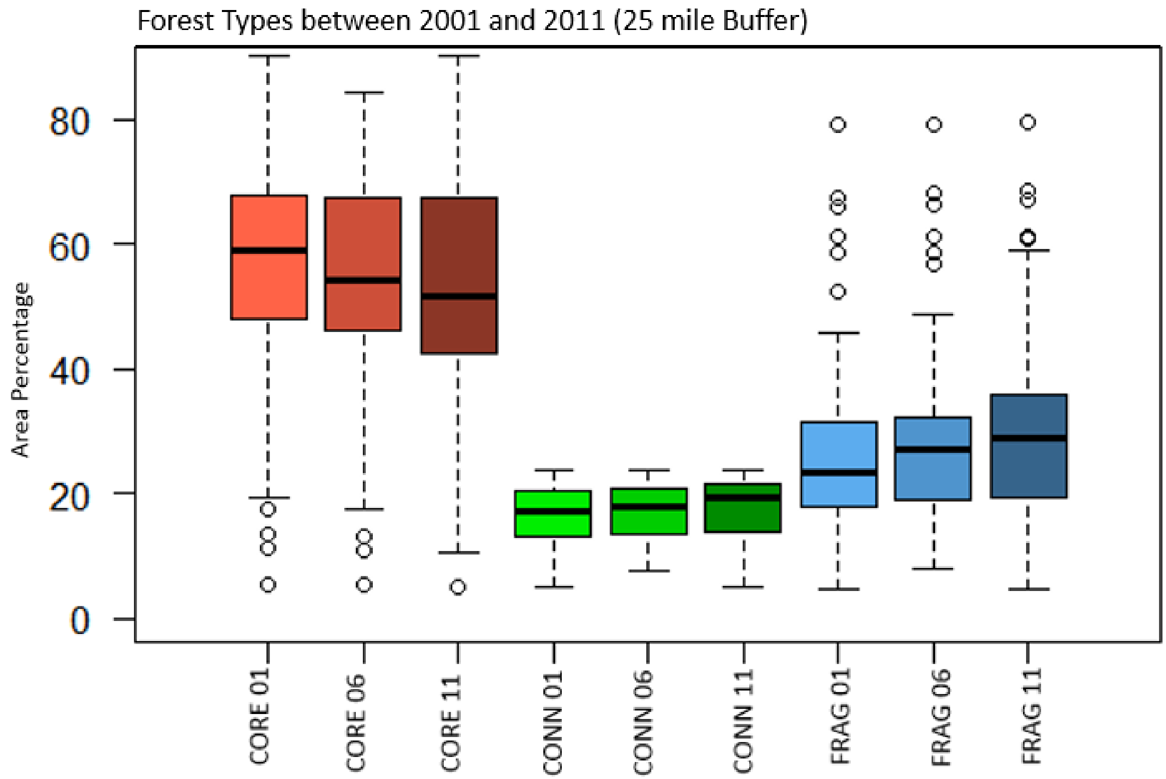


Figure 11. Changes in forest fragmentation in Virginia between 2001 and 2011 using a 25 mile buffer radius. While Core forest (red shades) declined between 2001 and 2011, both Connected (green shades) and Fragmented (blue shades) forest increased.

Tables

Table 1. Topographic slope categories used in this analysis.

Slope Categories	Slope Percentage (%)
1	0 – 0.5
2	> 0.5 - 2
3	> 2 - 5
4	> 5 - 9
5	> 9 - 15
6	> 15 - 30
7	> 30 - 60
8	Over 60

Table 2. Changes in forest types from 2001 and 2011 in the different buffer regions

BUFFER	FOREST TYPE	SIGNIFICANCE
1 mile	Core	0.7557
	Connected	0.7254
	Fragmented	0.7945
5 mile	Core	0.7318
	Connected	2.2e-16
	Fragmented	0.3778
15 mile	Core	0.6799
	Connected	2.2e-16
	Fragmented	5.503e-05
20 mile	Core	0.6644
	Connected	2.1e-16
	Fragmented	1.833e-06
25 mile	Core	0.5134
	Connected	2.3e-16
	Fragmented	2.431e-08

Table 3. Significant variables affecting Ovenbird populations in 1 mile buffer.

Model Variables	Posterior Mean	Lower 95% CI	Upper 95% CI	pMCMC
Intercept (2001)	2.312304	-3.61887	8.57661	0.447714
as.factor (2006)	-0.00619	-0.18303	0.166347	0.938571
as.factor (2011)	0.008696	-0.17462	0.201517	0.912571
Connected Forest	0.016077	-0.07289	0.113797	0.736571
Fragmented Forest	-0.05995	-0.09557	-0.01782	**0.002**
Blue Ridge	-0.00398	-0.01709	0.01071	0.564857
Coastal Plain	0.025478	0.003107	0.046312	**0.017429**
Piedmont	0.000837	-0.0157	0.020092	0.937429
Valley and Ridge	-0.00514	-0.02004	0.010505	0.489714
Slope1	0.024155	-0.02714	0.074963	0.343429
Slope2	-0.0481	-0.12879	0.036577	0.252
Slope4	0.065938	-0.06455	0.209657	0.336286
Slope5	-0.06706	-0.12555	-0.01046	**0.023143**
Slope6	0.012682	-0.08375	0.109229	0.792286
Slope7	0.033509	-0.02592	0.094083	0.263143
Slope8	-0.03242	-0.25165	0.185129	0.765143

Table 4. Significant variables affecting Wood thrush populations in 1 mile buffer.

Model Variables	Posterior Mean	Lower 95% CI	Upper 95% CI	pMCMC
Intercept (2001)	0.93577	-4.99489	6.933566	0.764
as.factor (2006)	-0.12754	-0.29067	0.047477	0.137143
as.factor (2011)	-0.5801	-0.75966	-0.37993	**0.000143**
Connected Forest	-0.07986	-0.17886	0.025016	0.130286
Fragmented Forest	0.017619	-0.02669	0.058456	0.413143
Blue Ridge	-0.00038	-0.01717	0.015061	0.960857
Coastal Plain	0.019287	-0.00554	0.04202	0.115143
Piedmont	0.007039	-0.01115	0.027924	0.472857
Valley and Ridge	0.005529	-0.01166	0.022874	0.507143
Slope1	-0.00413	-0.06304	0.053	0.897714
Slope3	-0.01805	-0.10938	0.0726	0.692857
Slope4	0.086443	-0.01114	0.192908	0.091714
Slope5	-0.04252	-0.14857	0.077147	0.454857
Slope6	0.02756	-0.04696	0.098633	0.457143
Slope7	0.013383	-0.06267	0.09279	0.726
Slope8	0.061436	-0.18213	0.285244	0.596571

Table 5. Significant variables affecting Ovenbird populations in 5 mile buffer.

Model Variables	**Posterior Mean**	**Lower 95% CI**	**Upper 95% CI**	**pMCMC**
Intercept (2001)	1.484353	-8.24956	10.85013	0.76171
as.factor (2006)	-0.01121	-0.18269	0.174739	0.9
as.factor (2011)	-0.01102	-0.22352	0.183804	0.91771
Connected Forest	0.028748	-0.08609	0.161252	0.65343
Fragmented Forest	-0.04882	-0.09575	-0.00208	**0.04486**
Blue Ridge	0.004797	-0.01596	0.025751	0.666
Coastal Plain	0.035169	0.007757	0.062695	**0.00743**
Piedmont	0.00395	-0.01841	0.028955	0.76571
Valley and Ridge	0.003563	-0.01886	0.024633	0.75629
Slope1	0.019131	-0.05742	0.098037	0.62514
Slope2	-0.07822	-0.24494	0.094051	0.36086
Slope4	0.099043	-0.11767	0.308045	0.36286
Slope5	-0.10699	-0.19288	-0.01996	**0.014**
Slope6	0.00806	-0.14285	0.157487	0.91314
Slope7	0.043097	-0.05303	0.14831	0.39686
Slope8	-0.01976	-0.34875	0.282534	0.91857

Table 6. Significant variables affecting Wood thrush populations in 5 mile buffer.

Model Variables	Posterior Mean	Lower 95% CI	Upper 95% CI	pMCMC
Intercept (2001)	4.125226	-5.62165	14.21436	0.41
as.factor (2006)	-0.13946	-0.32323	0.038706	0.134
as.factor (2011)	-0.52958	-0.73731	-0.31121	**<1e-04**
Connected Forest	-0.03348	-0.14655	0.081077	0.571
Fragmented Forest	-0.02275	-0.07091	0.02435	0.351
Blue Ridge	-0.00412	-0.02452	0.016293	0.681
Coastal Plain	0.007754	-0.02102	0.034935	0.583
Piedmont	0.000503	-0.02198	0.024129	0.96
Valley Ridge	0.001423	-0.02061	0.021977	0.898
Slope1	-0.0029	-0.11185	0.105045	0.955
Slope3	-0.05254	-0.21237	0.11144	0.512
Slope4	0.060702	-0.04611	0.172105	0.284
Slope5	-0.05155	-0.22741	0.126136	0.577
Slope6	-0.01197	-0.10921	0.083783	0.79
Slope7	-0.00373	-0.13764	0.127423	0.957
Slope8	-0.04415	-0.33095	0.229077	0.752

Table 7. Significant variables affecting Ovenbird populations in 15 mile buffer.

Model Variables	Posterior Mean	Lower 95% CI	Upper 95% CI	pMCMC
Intercept (2001)	3.467754	-10.0534	16.6905	0.6046
as.factor (2006)	-0.06856	-0.24851	0.098066	0.4509
as.factor (2011)	-0.15539	-0.36273	0.054095	0.1497
Connected Forest	0.084142	-0.06021	0.233762	0.2591
Fragmented Forest	-0.00302	-0.05952	0.055117	0.9191
Blue Ridge	0.01478	-0.02645	0.054716	0.4757
Coastal Plain	0.042148	-0.00211	0.086049	0.0657
Piedmont	0.018718	-0.0222	0.060505	0.3609
Valley and Ridge	0.022316	-0.01584	0.058204	0.24
Slope1	-0.06002	-0.16521	0.039055	0.2354
Slope2	-0.14053	-0.36105	0.081191	0.212
Slope4	-0.06048	-0.38022	0.244216	0.7097
Slope5	-0.04499	-0.18405	0.097581	0.5194
Slope6	-0.10631	-0.33892	0.12738	0.3674
Slope7	0.04625	-0.09564	0.190542	0.5174
Slope8	-0.10376	-0.41371	0.201941	0.4983

Table 8. Significant variables affecting Wood thrush populations in 15 mile buffer.

Model Variables	**Posterior Mean**	**Lower 95% CI**	**Upper 95% CI**	**pMCMC**
Intercept (2001)	-0.4672	-10.064	9.424537	0.9337
as.factor (2006)	-0.15083	-0.33825	0.034535	0.1103
as.factor (2011)	-0.56545	-0.77371	-0.34914	**<1e-04**
Connected Forest	-0.02856	-0.14554	0.093548	0.644
Fragmented Forest	-0.00731	-0.05442	0.04058	0.7691
Blue Ridge	0.018125	-0.01233	0.049467	0.2457
Coastal Plain	0.02916	-0.0047	0.063506	0.0966
Piedmont	0.015938	-0.01562	0.046651	0.3057
Valley and Ridge	0.017606	-0.01184	0.045381	0.2283
Slope1	0.01662	-0.10633	0.133381	0.7937
Slope3	-0.02133	-0.19763	0.138442	0.802
Slope4	0.096409	-0.03033	0.231501	0.14
Slope5	-0.0371	-0.23824	0.154172	0.712
Slope6	-0.00238	-0.12029	0.111673	0.9651
Slope7	0.046688	-0.08547	0.180339	0.5011
Slope8	0.015631	-0.18666	0.200825	0.8697

Table 9. Significant variables affecting Ovenbird populations in 20 mile buffer.

Model Variables	Posterior Mean	Lower 95% CI	Upper 95% CI	pMCMC
Intercept (2001)	3.56E+00	-1.04E+01	1.77E+01	0.61
as.factor (2006)	-6.74E-02	-2.38E-01	1.09E-01	0.454
as.factor (2011)	-1.40E-01	-3.37E-01	7.45E-02	0.187
Connected Forest	9.42E-02	-3.53E-02	2.14E-01	0.135
Fragmented Forest	-8.80E-03	-5.25E-02	3.52E-02	0.693
Blue Ridge	1.95E-04	-4.04E-02	3.82E-02	0.984
Coastal Plain	2.64E-02	-1.82E-02	7.18E-02	0.253
Piedmont	9.10E-03	-3.46E-02	5.22E-02	0.674
Valley and Ridge	9.54E-03	-2.36E-02	4.27E-02	0.559
Slope1	-4.86E-02	-1.56E-01	5.36E-02	0.357
Slope2	-1.03E-01	-3.45E-01	1.25E-01	0.381
Slope4	-6.79E-02	-3.56E-01	2.49E-01	0.653
Slope5	-8.27E-03	-1.33E-01	1.11E-01	0.891
Slope6	-1.19E-01	-3.37E-01	1.23E-01	0.299
Slope7	1.05E-01	-5.57E-02	2.81E-01	0.213
Slope8	-3.30E-01	-8.24E-01	1.68E-01	0.193

Table 10. Significant variables affecting Wood thrush populations in 20 mile buffer.

Model Variables	**Posterior Mean**	**Lower 95% CI**	**Upper 95% CI**	**pMCMC**
Intercept (2001)	-1.78E+00	-1.08E+01	7.49E+00	0.694
as.factor (2006)	-1.55E-01	-3.31E-01	3.68E-02	0.0971
as.factor (2011)	-5.74E-01	-7.78E-01	-3.53E-01	**<1e-04**
Connected Forest	-2.78E-02	-1.35E-01	8.17E-02	0.6089
Fragmented Forest	-4.90E-03	-4.25E-02	3.52E-02	0.8037
Blue Ridge	-3.64E-04	-2.81E-02	2.98E-02	0.9737
Coastal Plain	8.92E-03	-2.27E-02	3.87E-02	0.5629
Piedmont	9.88E-04	-2.81E-02	3.16E-02	0.952
Valley and Ridge	1.44E-03	-2.29E-02	2.59E-02	0.8983
Slope1	6.26E-02	-5.30E-02	1.83E-01	0.2954
Slope3	7.81E-02	-9.27E-02	2.55E-01	0.3646
Slope4	-1.47E-03	-1.01E-01	9.86E-02	0.9786
Slope5	1.06E-01	-7.30E-02	2.93E-01	0.26
Slope6	-7.62E-03	-1.07E-01	8.83E-02	0.8611
Slope7	1.03E-01	-6.20E-02	2.66E-01	0.2051
Slope8	-1.45E-01	-4.73E-01	1.72E-01	0.368

Table 11. Significant variables affecting Ovenbird populations in 25 mile buffer.

Model Variables	**Posterior Mean**	**Lower 95% CI**	**Upper 95% CI**	**pMCMC**
Intercept (2001)	7.177702	-3.11725	18.50138	0.196
as.factor (2006)	-0.09856	-0.27941	0.083868	0.2963
as.factor (2011)	-0.19009	-0.39012	0.019169	0.0706
Connected Forest	0.078469	-0.02307	0.18533	0.136
Fragmented Forest	0.003198	-0.03549	0.043324	0.874
Blue Ridge	-0.01595	-0.04746	0.017702	0.336
Coastal Plain	0.005291	-0.02798	0.038473	0.7429
Piedmont	-0.01234	-0.04632	0.021749	0.4694
Valley and Ridge	0.001566	-0.02575	0.029716	0.9109
Slope1	-0.07414	-0.15675	0.008259	0.0791
Slope2	-0.12348	-0.29995	0.060605	0.1794
Slope4	-0.07567	-0.3133	0.177085	0.5346
Slope5	-0.08293	-0.19051	0.017675	0.1111
Slope6	-0.05211	-0.21792	0.110791	0.5263
Slope7	-0.04928	-0.15866	0.062672	0.3829
Slope8	-0.0496	-0.16776	0.070577	0.4066

Table 12. Significant variables affecting Wood thrush populations in 25 mile buffer.

Model Variables	**Posterior Mean**	**Lower 95% CI**	**Upper 95% CI**	**pMCMC**
Intercept (2001)	1.785281	-5.28637	8.534741	0.6106
as.factor (2006)	-0.17731	-0.35528	0.011997	0.0671
as.factor (2011)	-0.62222	-0.83117	-0.40189	**<1e-04**
Connected Forest	-0.01035	-0.10529	0.083357	0.8389
Fragmented Forest	0.011201	-0.02224	0.047492	0.516
Blue Ridge	0.005236	-0.02046	0.030702	0.6911
Coastal Plain	0.00516	-0.02083	0.03188	0.698
Piedmont	-0.00091	-0.028	0.025485	0.9437
Valley and Ridge	0.006333	-0.01654	0.028496	0.5843
Slope1	-0.00302	-0.09396	0.090905	0.9566
Slope3	-0.0251	-0.14994	0.102159	0.6934
Slope4	0.046048	-0.04494	0.146752	0.3377
Slope5	-0.02131	-0.1627	0.113956	0.7503
Slope6	-0.00197	-0.08227	0.078213	0.9571
Slope7	0.003322	-0.08225	0.086437	0.9354
Slope8	-0.00856	-0.07652	0.066563	0.8091

Appendix D

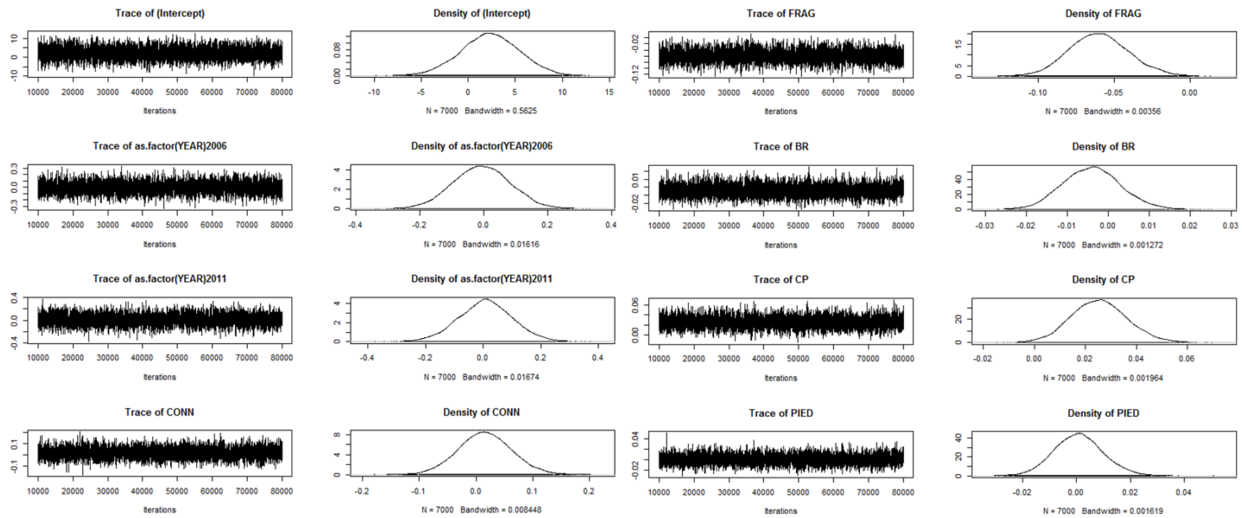
Appendix Di: R Code for determining the significance of the changes in forests in the buffer regions

```
buffer <- read.csv(f[f=='Buffer5.csv'],header=T)
buffer2 <- read.csv('Buffer5_transposed.csv',header=T)
# Get the names and columns of the data--
t(matrix(names(buffer))) ## makes a list

# Get repeated items such as route, geography, and slopes:
sel <- as.vector( sapply(1:70, function(x) rep(x,3)) )
buffer2 <- cbind.data.frame( buffer[sel,c(1,17:29)] ) ## cbind means column bind
View(buffer2) ## notice the row names looks weird
row.names(buffer2) <- 1:nrow(buffer2) ## rename the rows

# Add the year:
buffer2$YEAR <- rep(c(2001,2006,2011),70)

# Add the OV bird counts:
ov <- t(buffer[,4:6])
buffer2$OVCOUNT <- as.vector(ov)

# Add the WT bird counts:
wt <- t(buffer[,7:9])
buffer2$WTCOUNT <- as.vector(wt)

# Add the forest types:
ft <- t(buffer[,c(10,13,16)])
buffer2$CORE <- as.vector(ft)
ft <- t(buffer[,c(11,14,17)])
buffer2$CONN <- as.vector(ft)
ft <- t(buffer[,c(12,15,18)])
buffer2$FRAG <- as.vector(ft)

# Reorganize the columns:
t(matrix(names(buffer2)))
buffer2 <- buffer2[,c(1,15:20,2:14)]
View(buffer2)

# Clean up:
rm(wt,ft,ov,sel,f)

#-----------------------------------------------------------------
# Get group means:
#-----------------------------------------------------------------
apply(buffer[,c(5:7)],2,mean,na.rm=T) ## OV
apply(buffer[,c(8:10)],2,mean,na.rm=T) ## WT

apply(buffer[,c(10,13,16)],2,mean,na.rm=T) ## FRAG
apply(buffer[,c(8,11,14)],2,mean,na.rm=T) ## CORE
apply(buffer[,c(9,12,15)],2,mean,na.rm=T) ## CONN

#-----------------------------------------------------------------
# More Plots to Understand Change over Time:
#-----------------------------------------------------------------
par(mar=c(4,4,1,1))
boxplot(buffer2$OVCOUNT~buffer2$YEAR,las=2,xlim=c(0,7),at=1:3,
        names=c("OV01","OV06","OV11"), col=c('tomato','tomato3','tomato4'))
boxplot(buffer2$WTCOUNT~buffer2$YEAR,add=TRUE,at=4:6,axes=F,
        col=c('steelblue2','steelblue3','steelblue4'))
axis(1, at=4:6, labels=c("WT01","WT06","WT11"),las=2)

par(mar=c(4,4,1,1))
boxplot(buffer2$CORE~buffer2$YEAR,las=1,xlim=c(0,10),ylim=c(0,max(buffer2$CORE)),
        at=1:3, names=c("","CORE",""), col=c('tomato','tomato3','tomato4'))
boxplot(buffer2$FRAG~buffer2$YEAR,add=TRUE,at=7:9,axes=F,
        col=c('steelblue2','steelblue3','steelblue4'))
axis(1, at=7:9, labels=c("","FRAG",""))
boxplot(buffer2$CONN~buffer2$YEAR,add=TRUE,at=4:6,axes=F,
```

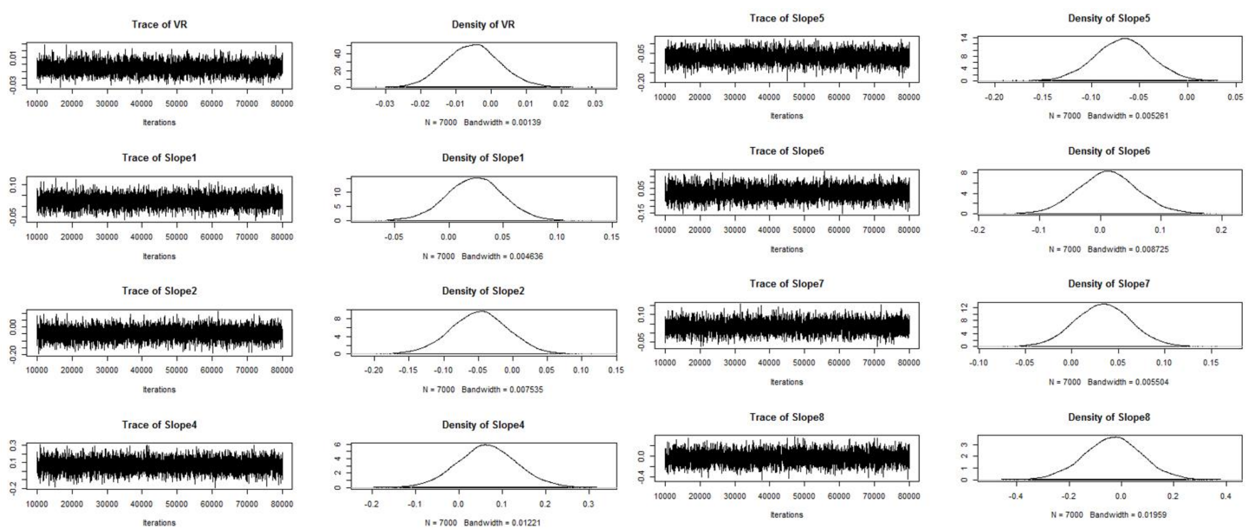
Appendix Dii: R Code used for Repeated Measures ANOVA

```
# Set working directory--
setwd("E:/IF/Ranalysis")

# See what files are in this directory--
list.files()
f <- list.files(pattern='.csv') ## looks for files that are csv

# Load in the data--
buffer <- read.csv(f[f=='Buffer25.csv'],header=T)

# Load in the data with transposed structure--
buffer2 <- read.csv(f[f=='Buffer25_transposed1.csv'],header=T)

# Get the names and columns of the data--
t(matrix(names(buffer)))
t(matrix(names(buffer2)))

#-------------------------------------------------------------------------------
# Clean Data for OV and WT-- remove NA's
#-------------------------------------------------------------------------------
buffer2$ROUTE[which(is.na(buffer2$OVCOUNT)==TRUE)]
OV_no_na <- buffer2[!( buffer2$ROUTE %in% c('BLACKSBURG','DRYDEN') ),]
OV_no_na <- OV_no_na[is.na(OV_no_na$OVCOUNT)==FALSE,]
which(is.na(OV_no_na$OVCOUNT)==T) ## double check NA's removed

buffer2$ROUTE[which(is.na(buffer2$WTCOUNT)==TRUE)]
WT_no_na <- buffer2[!( buffer2$ROUTE %in% c('PARTLOW','PORT ROYAL') ),]
WT_no_na <- WT_no_na[is.na(WT_no_na$WTCOUNT)==FALSE,]
which(is.na(WT_no_na$WTCOUNT)==T) ## double check NA's removed

#-------------------------------------------------------------------------------
# Using MCMCglmm (Markov Chain Monte Carlo Generalized Linear Mixed Model)
#-------------------------------------------------------------------------------
install.packages("MCMCglmm")
library(MCMCglmm)

# OV Population

## grab the dataset
dat <- OV_no_na

## run model
modOV <- MCMCglmm(OVCOUNT ~ as.factor(YEAR) + CONN + FRAG + BR + CP + PIED + VR +
                   Slope1 + Slope2 + Slope4 + Slope5 + Slope6 + Slope7 + Slope8,
                 random=~ROUTE, family = "poisson", data = dat,
                 prior = list(G = list(G1=list(V=1, nu=1)), R = list(V=1, nu=1)),
                 rcov=~ROUTE:YEAR,
                 verbose=F, pr=TRUE,
                 nitt=80000,burnin=10000,thin=10)

## get summary of output
summary(modOV)

## grab only effect estimates
summod <- summary(modOV)
summod$solutions

## traceplots of fixed effects
plot(modOV$Sol[,1:16])

## traceplots of random effects
plot(modOV$VCV)

# WT Population
#-------------------------------------------------------------
```

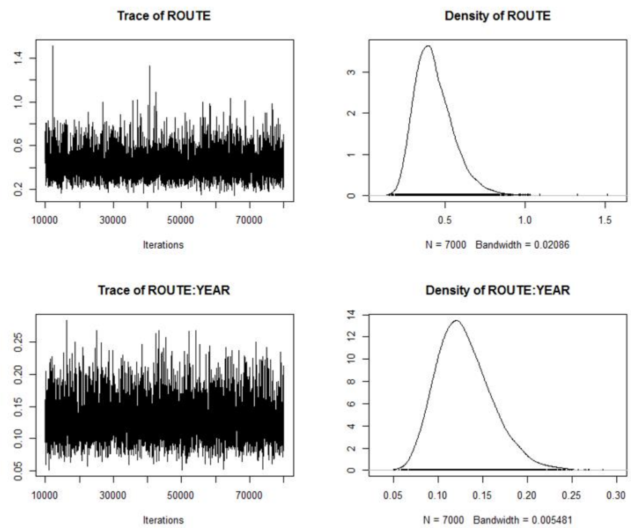
Appendix Diii. Traceplots for Ovenbird populations in 1 mile buffer



Appendix Div. Traceplots for Ovenbird populations in 1 mile buffer

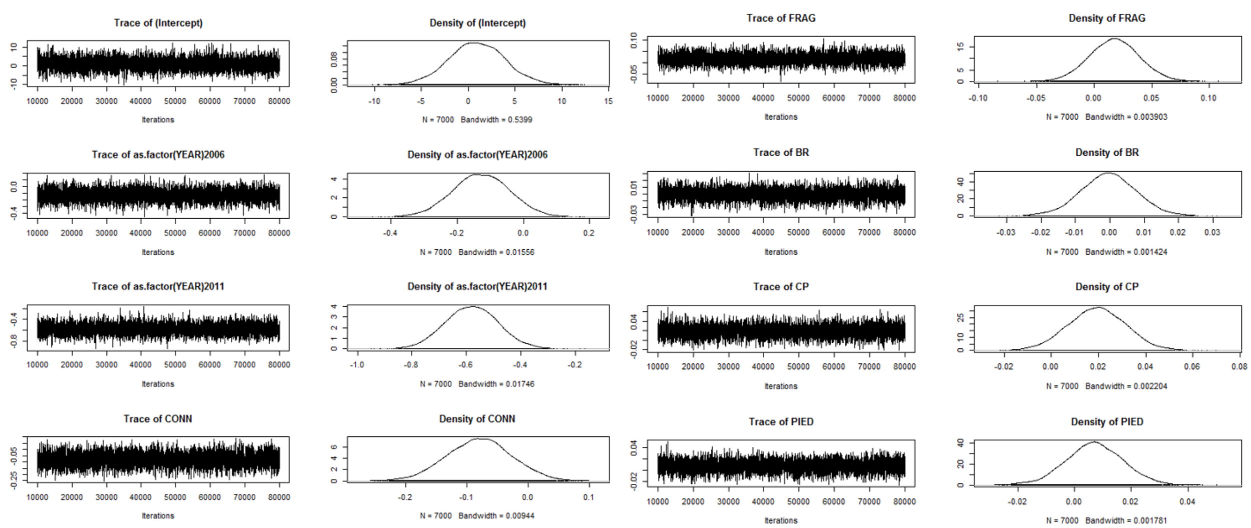


Appendix Dv. Traceplots for Ovenbird populations in 1 mile buffer and Model result summary

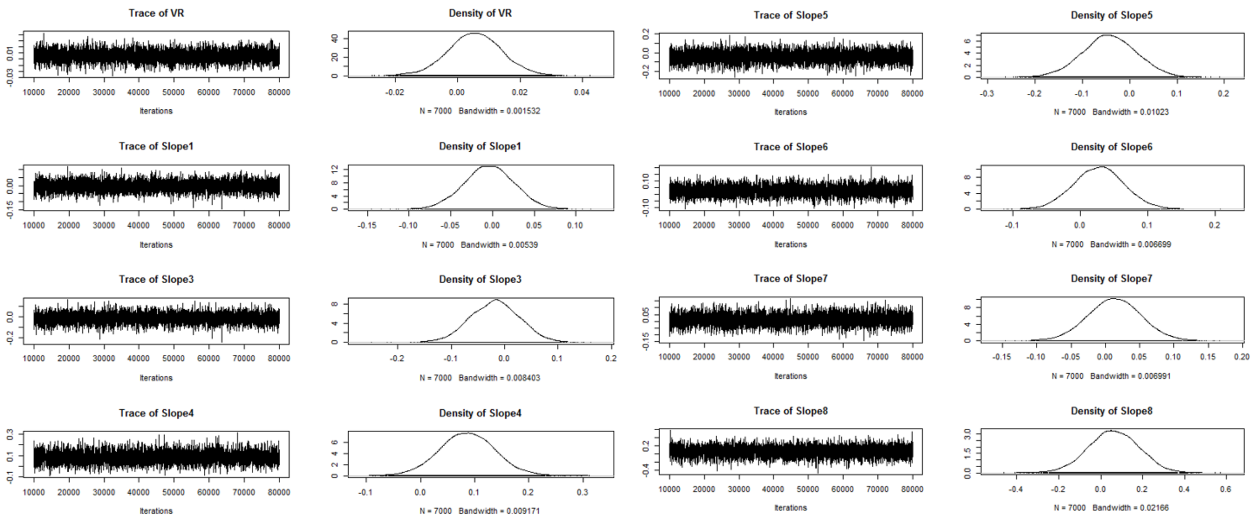



```
 Iterations = 10001:79991
 Thinning interval  = 10
 Sample size  = 7000

 DIC: 1091.785

 G-structure:  ~ROUTE

      post.mean l-95% CI u-95% CI eff.samp
ROUTE    0.4312   0.2224   0.6728     2688

 R-structure:  ~ROUTE:YEAR

           post.mean l-95% CI u-95% CI eff.samp
ROUTE:YEAR    0.1298   0.0751   0.1932     3222

 Location effects: OVCOUNT ~ as.factor(YEAR) + CONN + FRAG + BR + CP + PIED + VR + Slope1 + Slope2 + Slope4 + Slope5 + Slope6 +
 Slope7 + Slope8

                     post.mean   l-95% CI   u-95% CI eff.samp  pMCMC
(Intercept)          2.3123043 -3.6188732  8.5766100     6083 0.4477
as.factor(YEAR)2006 -0.0061871 -0.1830283  0.1663467     5576 0.9386
as.factor(YEAR)2011  0.0086963 -0.1746209  0.2015170     5628 0.9126
CONN                 0.0160769 -0.0728927  0.1137969     5758 0.7366
FRAG                -0.0599469 -0.0955668 -0.0178242     5113 0.0020 **
BR                  -0.0039776 -0.0170879  0.0107104     6118 0.5649
CP                   0.0254780  0.0031075  0.0463115     6271 0.0174 *
PIED                 0.0008369 -0.0157040  0.0200919     5766 0.9374
VR                  -0.0051448 -0.0200384  0.0105055     6271 0.4897
Slope1               0.0241547 -0.0271363  0.0749630     5507 0.3434
Slope2              -0.0480995 -0.1287914  0.0365767     5437 0.2520
Slope4               0.0659376 -0.0645512  0.2096572     6197 0.3363
Slope5              -0.0670620 -0.1255474 -0.0104629     4464 0.0231 *
Slope6               0.0126819 -0.0837516  0.1092290     5365 0.7923
Slope7               0.0335089 -0.0259186  0.0940828     6670 0.2631
Slope8              -0.0324188 -0.2516501  0.1851291     6293 0.7651
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
```

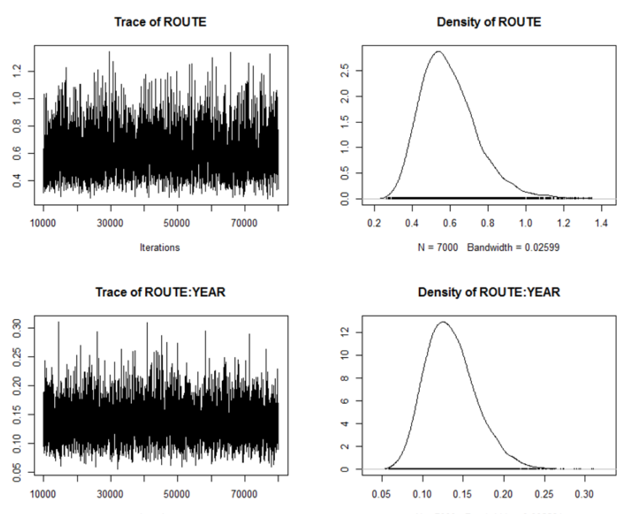
Appendix Dvi. Traceplots for Wood thrush populations in 1 mile buffer



Appendix Dvii. Traceplots for Wood thrush populations in 1 mile buffer

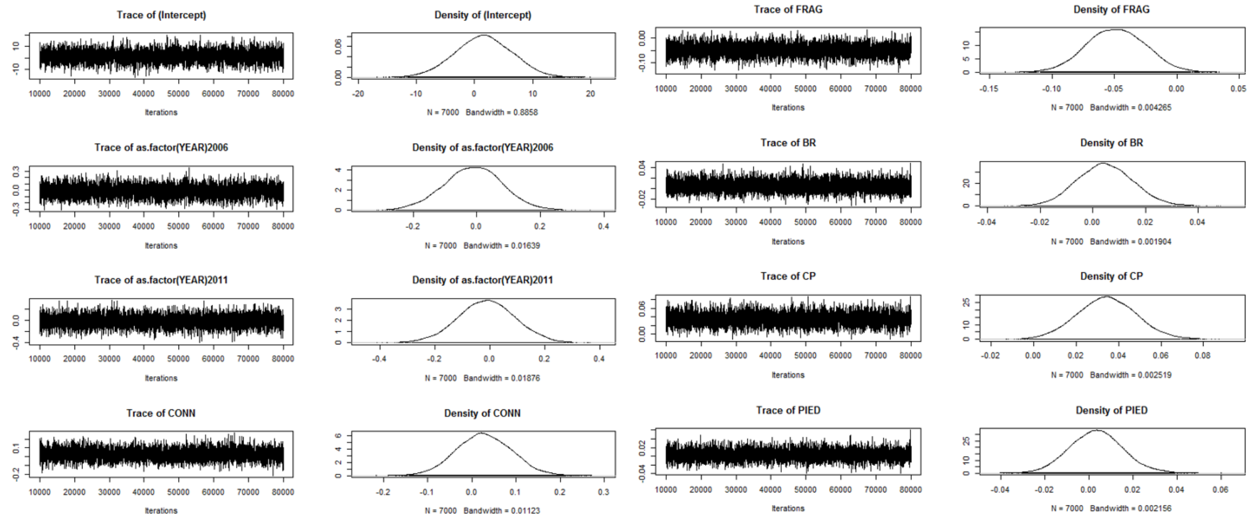


Appendix Dviii. Traceplots for Wood thrush populations in 1 mile buffer and Model Summary Results

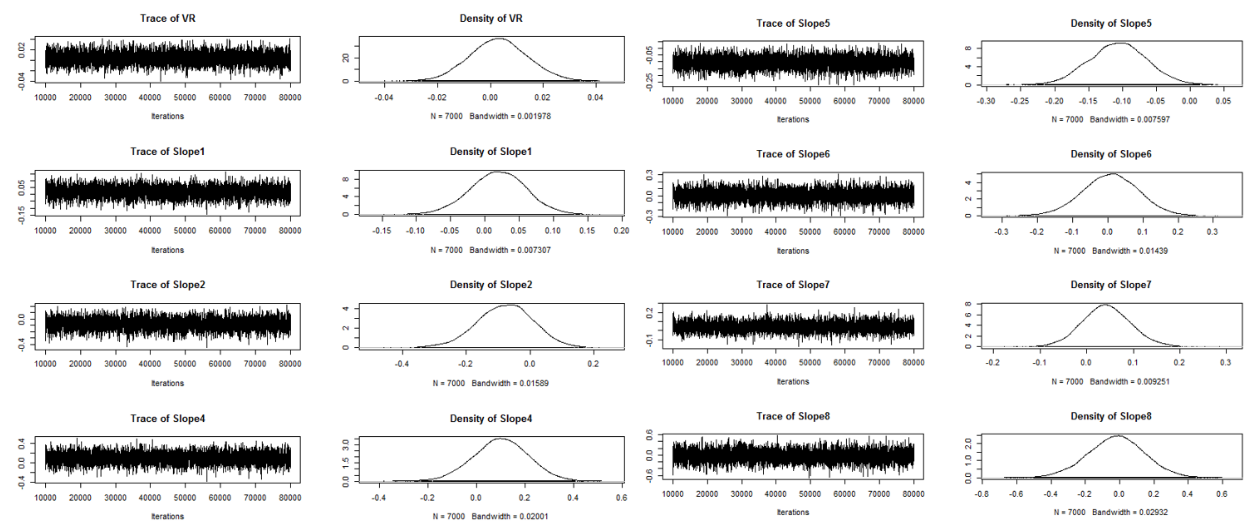



```
 Iterations = 10001:79991
 Thinning interval  = 10
 Sample size  = 7000

 DIC: 1104.891

 G-structure:  ~ROUTE

      post.mean l-95% CI u-95% CI eff.samp
ROUTE     0.594   0.3283   0.8924     3317

 R-structure:  ~ROUTE:YEAR

           post.mean l-95% CI u-95% CI eff.samp
ROUTE:YEAR    0.1353   0.0771   0.1967     3214

 Location effects: WTCOUNT ~ as.factor(YEAR) + CONN + FRAG + BR + CP + PIED + VR + Slope1 + Slope3 + Slope4 + Slope5 + Slope6 +
 Slope7 + Slope8

                     post.mean   l-95% CI   u-95% CI eff.samp  pMCMC
(Intercept)          0.9357701 -4.9948864  6.9335662     6704 0.7640
as.factor(YEAR)2006 -0.1275414 -0.2906736  0.0474773     6041 0.1371
as.factor(YEAR)2011 -0.5800985 -0.7596612 -0.3799311     5748 <1e-04 ***
CONN                -0.0798586 -0.1788635  0.0250161     6631 0.1303
FRAG                 0.0176187 -0.0266948  0.0584564     6495 0.4131
BR                  -0.0003793 -0.0171694  0.0150612     5963 0.9609
CP                   0.0192875 -0.0055374  0.0420195     6358 0.1151
PIED                 0.0070387 -0.0111543  0.0279236     5933 0.4729
VR                   0.0055293 -0.0116641  0.0228740     6519 0.5071
Slope1              -0.0041316 -0.0630353  0.0530000     7000 0.8977
Slope3              -0.0180534 -0.1093780  0.0725999     6697 0.6929
Slope4               0.0864426 -0.0111408  0.1929077     6256 0.0917 .
Slope5              -0.0425155 -0.1485731  0.0771469     7000 0.4549
Slope6               0.0275603 -0.0469633  0.0986327     5901 0.4571
Slope7               0.0133826 -0.0626663  0.0927900     7000 0.7260
Slope8               0.0614364 -0.1821323  0.2852437     7000 0.5966
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
```

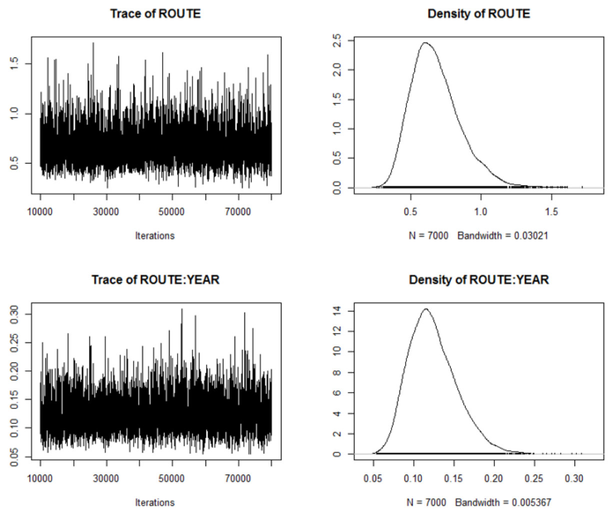
Appendix Dix. Traceplots for Ovenbird populations in 5 m buffer



Appendix Dx. Traceplots for Ovenbird populations in 5 m buffer

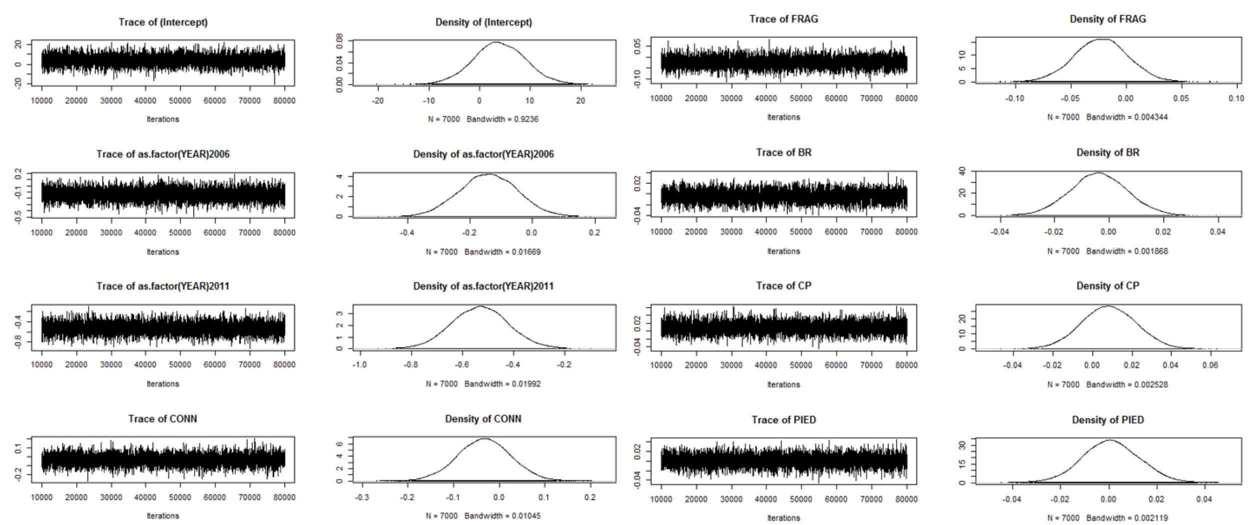


Appendix Dxi. Traceplots for Ovenbird populations in 5 m buffer and Model Summary Results

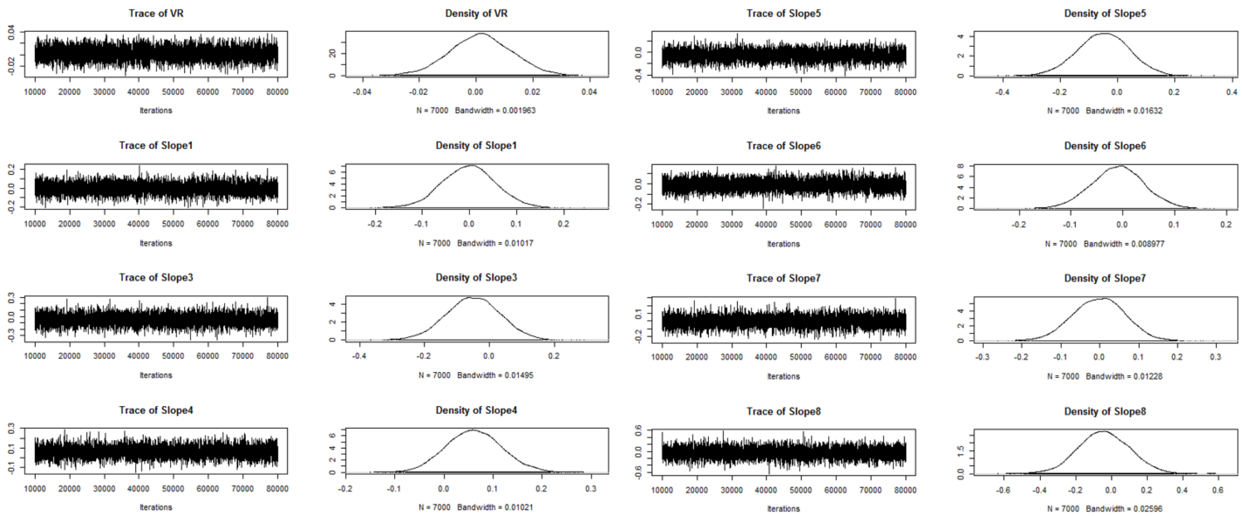



```
 Iterations = 10001:79991
 Thinning interval  = 10
 Sample size  = 7000

 DIC: 1090.876

 G-structure:  ~ROUTE

      post.mean l-95% CI u-95% CI eff.samp
ROUTE    0.6793   0.3852    1.045     2736

 R-structure:  ~ROUTE:YEAR

           post.mean l-95% CI u-95% CI eff.samp
ROUTE:YEAR    0.1245   0.0742   0.1877     3369

 Location effects: OVCOUNT ~ as.factor(YEAR) + CONN + FRAG + BR + CP + PIED + VR + Slope1 + Slope2 + Slope4 + Slope5 + Slope6 +
 Slope7 + Slope8

                    post.mean  l-95% CI  u-95% CI eff.samp  pMCMC
(Intercept)          1.552917 -8.263507 11.020883     5752 0.7471
as.factor(YEAR)2006 -0.011405 -0.193415  0.164428     5473 0.9140
as.factor(YEAR)2011 -0.011901 -0.211123  0.202412     5728 0.9077
CONN                 0.027935 -0.093349  0.149766     5260 0.6591
FRAG                -0.048704 -0.093960 -0.002031     5554 0.0403 *
BR                   0.004692 -0.015618  0.025498     6634 0.6637
CP                   0.034576  0.007124  0.062560     6294 0.0123 *
PIED                 0.003449 -0.020869  0.026330     6119 0.7726
VR                   0.003213 -0.017994  0.025156     6461 0.7757
Slope1               0.018921 -0.060039  0.100506     4885 0.6326
Slope2              -0.077949 -0.247102  0.100846     5348 0.3760
Slope4               0.098770 -0.130581  0.310203     5952 0.3791
Slope5              -0.106687 -0.189802 -0.021053     6047 0.0146 *
Slope6               0.007600 -0.143676  0.169858     5838 0.9109
Slope7               0.042385 -0.057248  0.148908     5314 0.4080
Slope8              -0.019870 -0.345973  0.307451     6700 0.9171
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
```

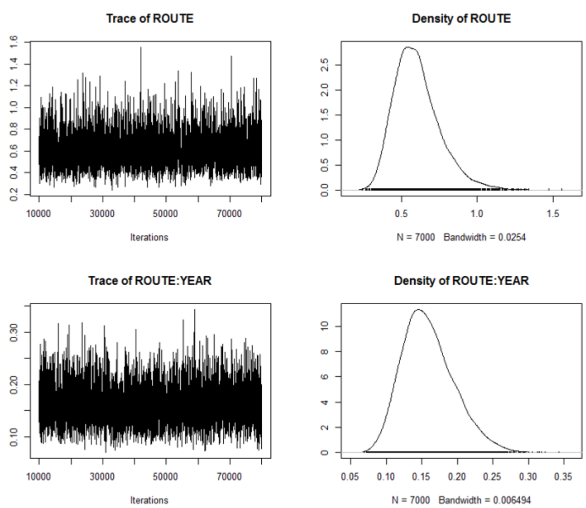
Appendix Dxii. Traceplots for Wood thrush populations in 5 m buffer



Appendix Dxiii. Traceplots for Wood thrush populations in 5 m buffer

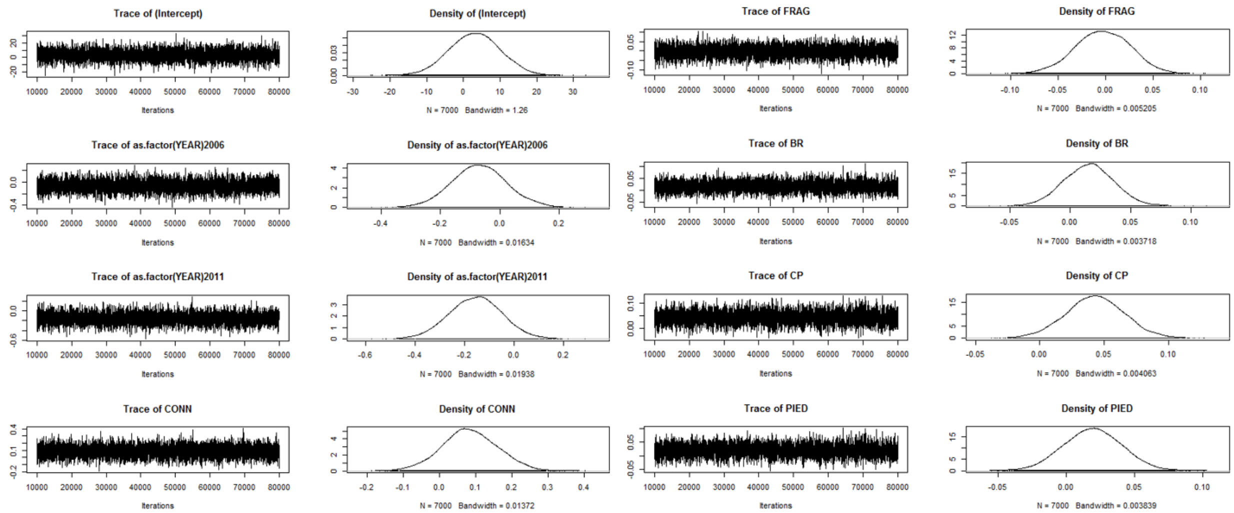


Appendix Dxiv. Traceplots for Wood thrush populations in 5 m buffer and Model Summary Results

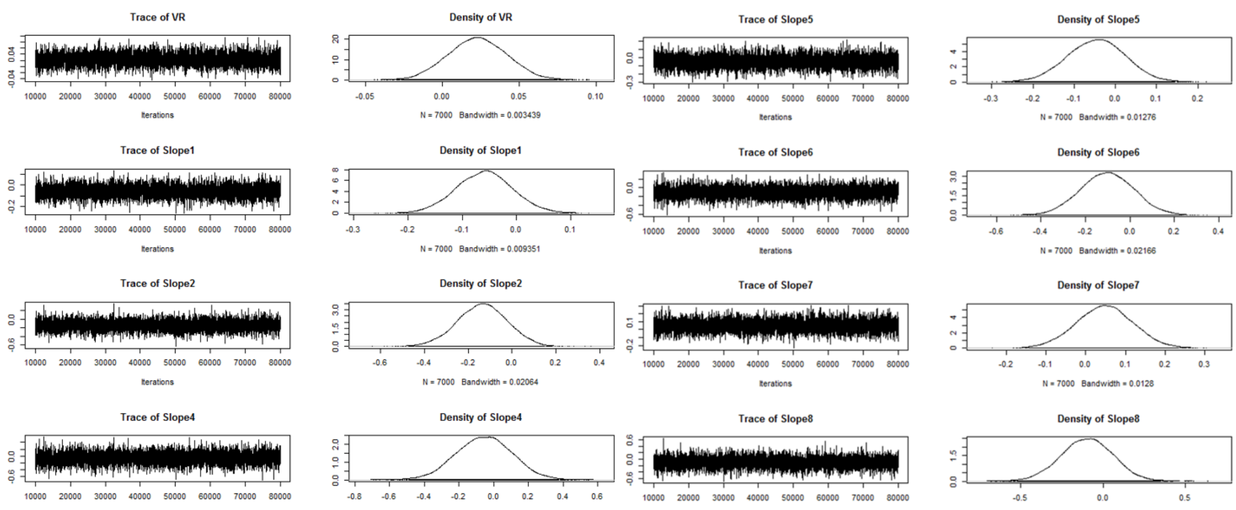



```
 Iterations = 10001:79991
 Thinning interval  = 10
 Sample size  = 7000

 DIC: 1110.301

 G-structure:  ~ROUTE

      post.mean l-95% CI u-95% CI eff.samp
ROUTE    0.6045   0.3414   0.9106     4077

 R-structure:  ~ROUTE:YEAR

           post.mean l-95% CI u-95% CI eff.samp
ROUTE:YEAR    0.1594  0.09201   0.2318     3471

 Location effects: WTCOUNT ~ as.factor(YEAR) + CONN + FRAG + BR + CP + PIED + VR + Slope1 + Slope3 + Slope4 + Slope5 + Slope6 +
 Slope7 + Slope8

                    post.mean   l-95% CI   u-95% CI eff.samp  pMCMC
(Intercept)         3.9217261 -6.1368107 14.0893397     6137  0.435
as.factor(YEAR)2006 -0.1412554 -0.3207363  0.0387097     6157  0.127
as.factor(YEAR)2011 -0.5309900 -0.7466843 -0.3122915     6451 <1e-04 ***
CONN               -0.0330024 -0.1482397  0.0806614     6648  0.567
FRAG               -0.0225906 -0.0692454  0.0259798     5880  0.348
BR                 -0.0038903 -0.0242284  0.0168195     6096  0.707
CP                  0.0082884 -0.0194783  0.0349811     7000  0.553
PIED                0.0008927 -0.0212589  0.0252881     6518  0.948
VR                  0.0016987 -0.0197690  0.0230549     6645  0.869
Slope1             -0.0010076 -0.1101323  0.1128244     6705  0.995
Slope3             -0.0499266 -0.2118135  0.1135460     5490  0.546
Slope4              0.0612189 -0.0453653  0.1762295     7000  0.277
Slope5             -0.0496820 -0.2309251  0.1249889     6713  0.589
Slope6             -0.0098284 -0.1055048  0.0918273     6364  0.851
Slope7             -0.0024026 -0.1365975  0.1276088     6282  0.985
Slope8             -0.0398816 -0.3087364  0.2519364     7000  0.766
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
```

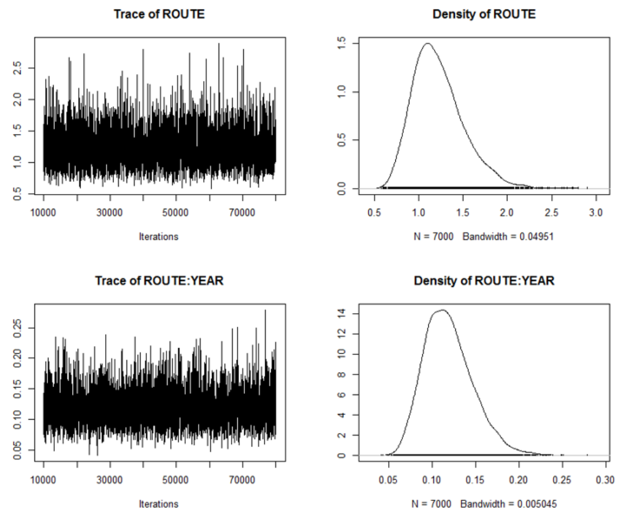
Appendix Dxv. Traceplots for Ovenbird populations in 15 m buffer



Appendix Dxvi. Traceplots for Ovenbird populations in 15 m buffer

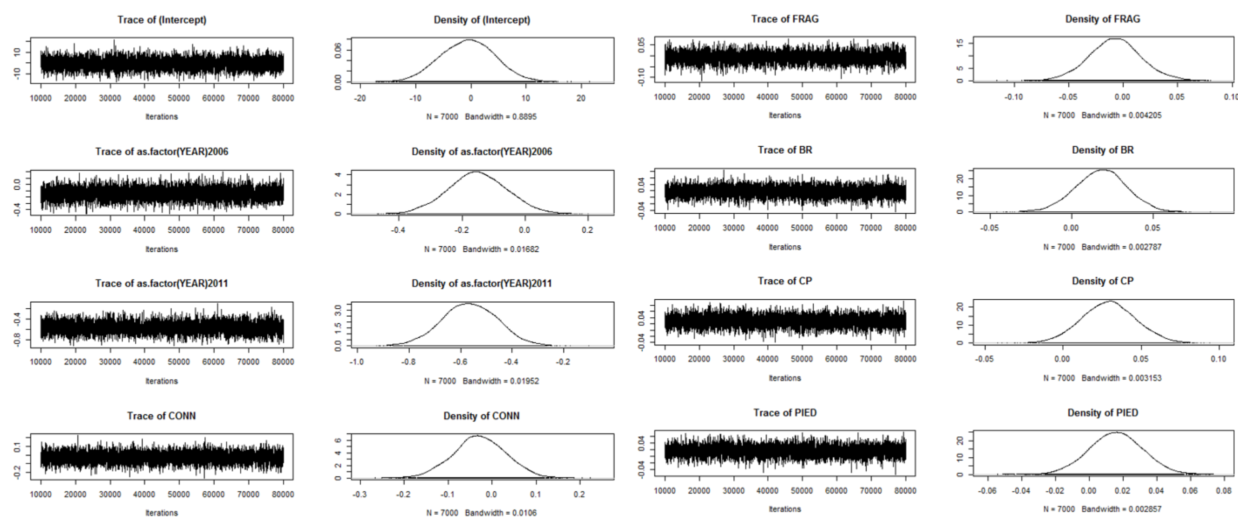


Appendix Dxvii. Traceplots for Ovenbird populations in 15 m buffer and Model Summary Results

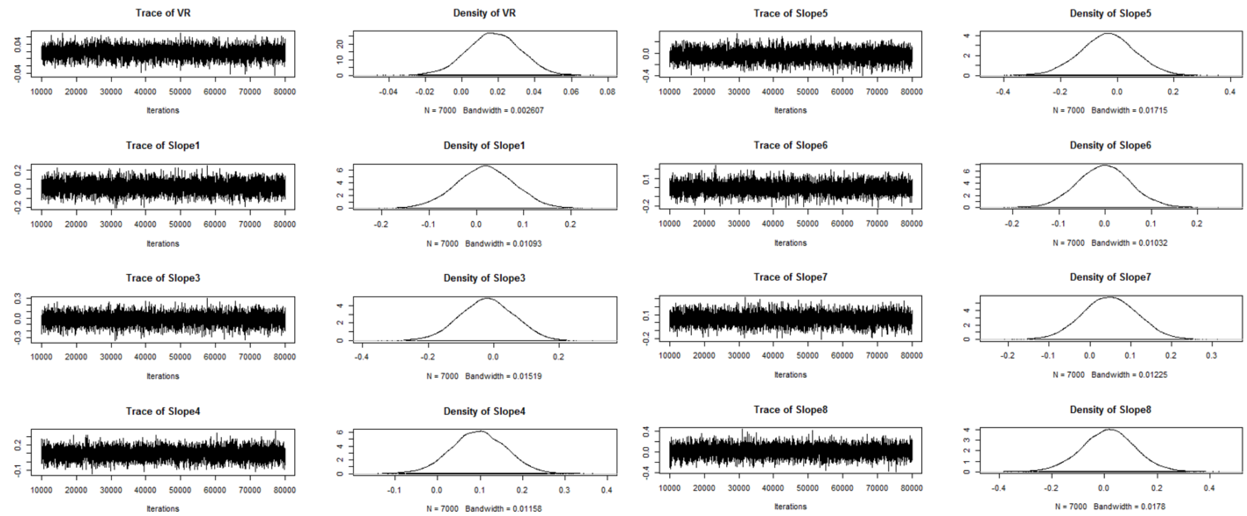



```
 Iterations = 10001:79991
 Thinning interval  = 10
 Sample size  = 7000

 DIC: 1087.516

 G-structure:  ~ROUTE

      post.mean l-95% CI u-95% CI eff.samp
ROUTE     1.225   0.7162    1.797     2730

 R-structure:  ~ROUTE:YEAR

           post.mean l-95% CI u-95% CI eff.samp
ROUTE:YEAR    0.1194  0.06706   0.1789     3174

 Location effects: OVCOUNT ~ as.factor(YEAR) + CONN + FRAG + BR + CP + PIED + VR + Slope1 + Slope2 + Slope4 + Slope5 + Slope6 +
 Slope7 + Slope8

                    post.mean   l-95% CI   u-95% CI eff.samp pMCMC
(Intercept)          3.406016 -10.353041 16.923530     5961 0.625
as.factor(YEAR)2006 -0.067815  -0.236906  0.112670     5821 0.446
as.factor(YEAR)2011 -0.155865  -0.362842  0.049773     6185 0.135
CONN                 0.084264  -0.061822  0.240268     6276 0.270
FRAG                -0.002816  -0.057526  0.054276     6427 0.930
BR                   0.014697  -0.027647  0.052616     7000 0.480
CP                   0.042498  -0.002630  0.087251     6420 0.062 .
PIED                 0.018979  -0.021408  0.061039     6478 0.372
VR                   0.022386  -0.013722  0.061205     6686 0.235
Slope1              -0.059613  -0.159225  0.039181     5732 0.243
Slope2              -0.140937  -0.353803  0.092899     5181 0.215
Slope4              -0.058468  -0.380958  0.245678     5817 0.709
Slope5              -0.047125  -0.182478  0.097599     7000 0.491
Slope6              -0.104872  -0.343762  0.115514     5830 0.370
Slope7               0.046873  -0.097885  0.185965     5885 0.522
Slope8              -0.103701  -0.406055  0.199426     6751 0.503
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
```

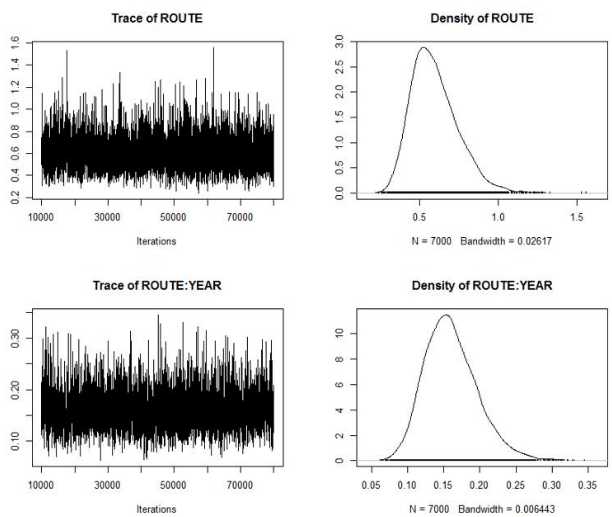
Appendix Dxviii. Traceplots for Wood thrush populations in 15 m buffer



Appendix Dxix. Traceplots for Wood thrush populations in 15 m buffer

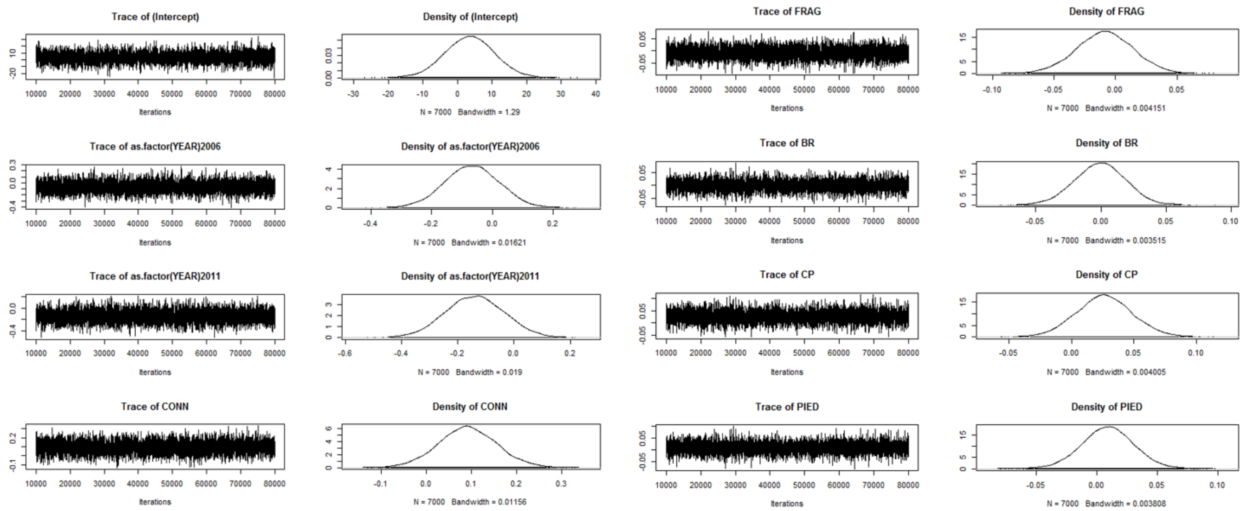


Appendix Dxx. Traceplots for Wood thrush populations in 15 m buffer and Model Summary Results

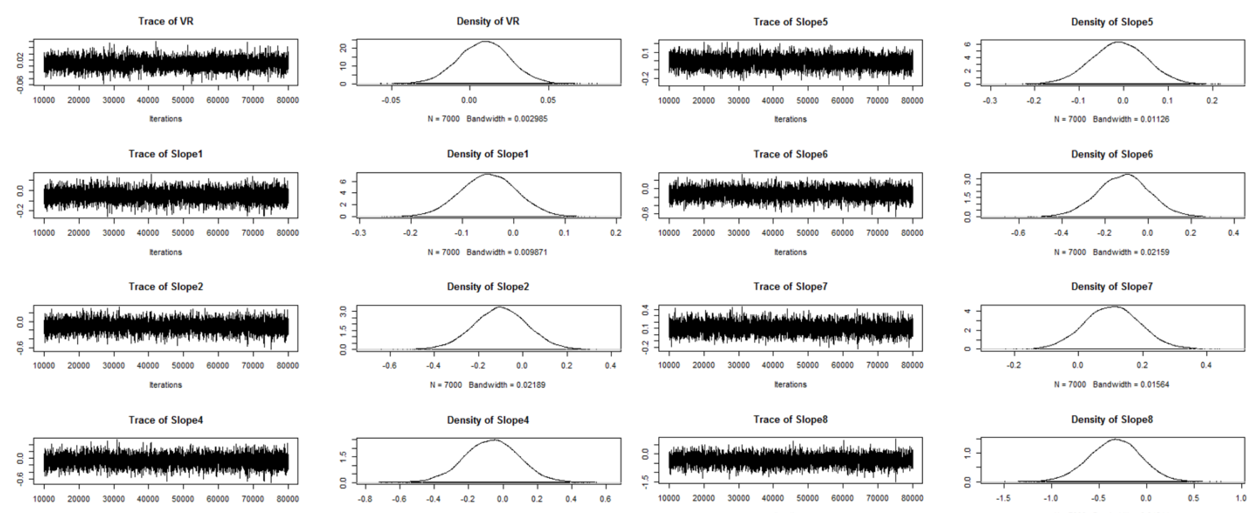



```
 Iterations = 10001:79991
 Thinning interval  = 10
 Sample size  = 7000

 DIC: 1108.814

 G-structure:  ~ROUTE

      post.mean l-95% CI u-95% CI eff.samp
ROUTE    0.5975    0.333   0.8854     3814

 R-structure:  ~ROUTE:YEAR

           post.mean l-95% CI u-95% CI eff.samp
ROUTE:YEAR    0.1605  0.09312   0.2347     3302

 Location effects: WTCOUNT ~ as.factor(YEAR) + CONN + FRAG + BR + CP + PIED + VR + Slope1 + Slope3 + Slope4 + Slope5 + Slope6 +
 Slope7 + Slope8

                   post.mean   l-95% CI   u-95% CI eff.samp  pMCMC
(Intercept)        -0.567228 -10.472803   8.645246     6089 0.9189
as.factor(YEAR)2006 -0.150559  -0.341699   0.025401     6474 0.1120
as.factor(YEAR)2011 -0.564803  -0.785655  -0.360498     6035 <1e-04 ***
CONN               -0.028242  -0.148935   0.083939     6641 0.6389
FRAG               -0.007470  -0.056383   0.040209     5876 0.7480
BR                  0.018043  -0.012090   0.048685     6179 0.2474
CP                  0.029227  -0.003762   0.064901     6339 0.0929 .
PIED                0.015964  -0.015585   0.046050     6121 0.3089
VR                  0.017512  -0.010080   0.046695     6057 0.2283
Slope1              0.018316  -0.103709   0.133247     6422 0.7480
Slope3             -0.019772  -0.183724   0.143909     6054 0.8109
Slope4              0.096389  -0.026431   0.223665     6469 0.1306
Slope5             -0.034639  -0.220426   0.163774     6565 0.7160
Slope6             -0.002386  -0.111657   0.115692     6534 0.9666
Slope7              0.048874  -0.079176   0.185521     6069 0.4677
Slope8              0.014141  -0.183662   0.214489     6520 0.8737
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
```

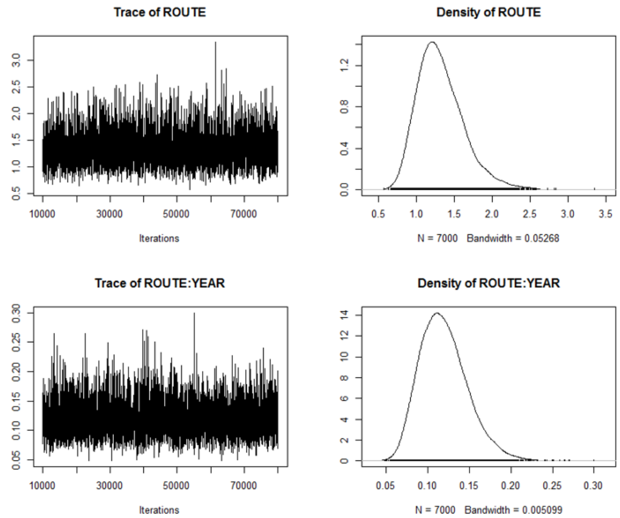
Appendix Dxxi. Traceplots for Ovenbird populations in 20 m buffer



Appendix Dxxii. Traceplots for Ovenbird populations in 20 m buffer

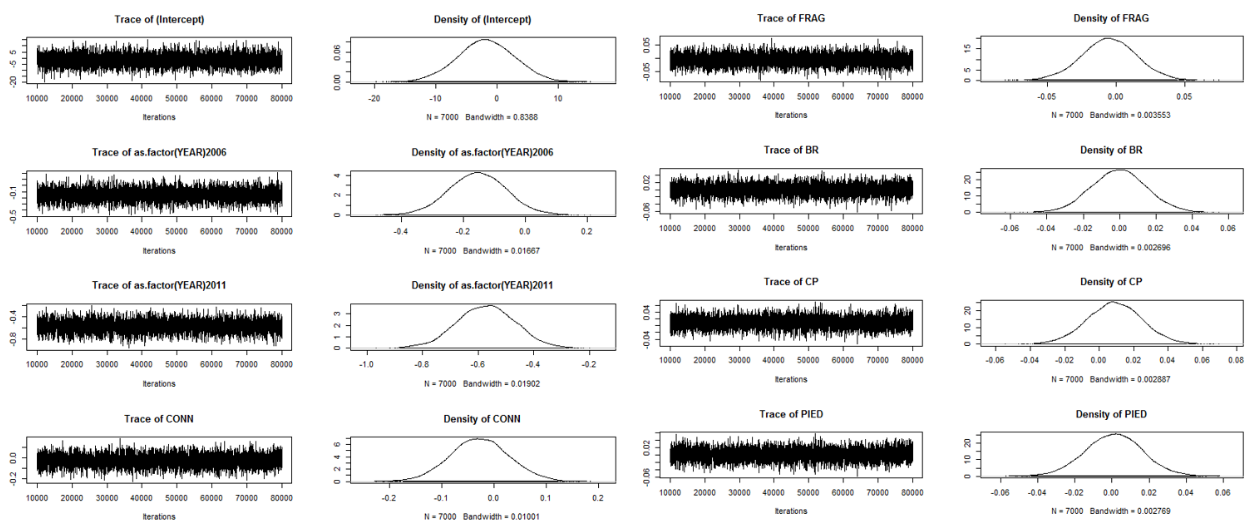


Appendix Dxxiii. Traceplots for Ovenbird populations in 20 m buffer and Model Summary Results

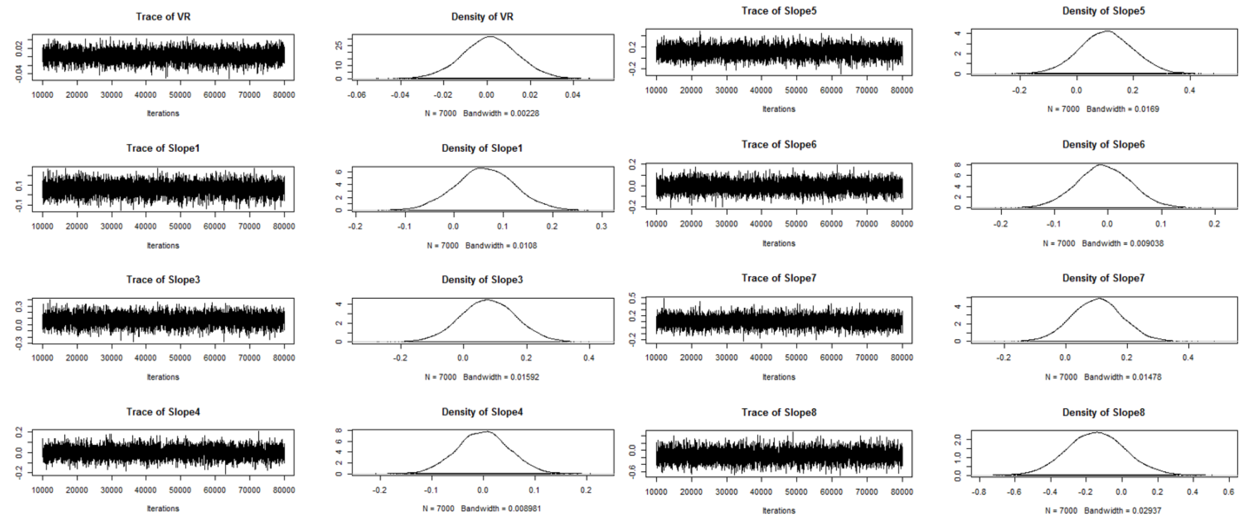



```
 Iterations = 10001:79991
 Thinning interval  = 10
 Sample size  = 7000

 DIC: 1087.441

 G-structure:  ~ROUTE

      post.mean l-95% CI u-95% CI eff.samp
ROUTE     1.328   0.8056    1.949     2260

 R-structure:  ~ROUTE:YEAR

           post.mean l-95% CI u-95% CI eff.samp
ROUTE:YEAR    0.1201  0.06924   0.1768     3092

 Location effects: OVCOUNT ~ as.factor(YEAR) + CONN + FRAG + BR + CP + PIED + VR + Slope1 + Slope2 + Slope4 + Slope5 + Slope6 +
 Slope7 + Slope8

                     post.mean   l-95% CI   u-95% CI eff.samp pMCMC
(Intercept)          3.579e+00 -1.037e+01  1.771e+01     5335 0.601
as.factor(YEAR)2006 -6.505e-02 -2.375e-01  1.132e-01     5527 0.465
as.factor(YEAR)2011 -1.400e-01 -3.421e-01  6.211e-02     5952 0.179
CONN                 9.275e-02 -2.721e-02  2.281e-01     5820 0.154
FRAG                -9.248e-03 -5.276e-02  3.694e-02     7000 0.686
BR                  -8.421e-05 -3.720e-02  3.976e-02     7000 0.992
CP                   2.655e-02 -1.661e-02  7.260e-02     7000 0.241
PIED                 9.040e-03 -3.422e-02  5.169e-02     7000 0.672
VR                   9.275e-03 -2.468e-02  4.051e-02     7000 0.567
Slope1              -4.822e-02 -1.595e-01  5.109e-02     5407 0.360
Slope2              -1.034e-01 -3.562e-01  1.207e-01     5146 0.395
Slope4              -6.580e-02 -3.836e-01  2.275e-01     5375 0.656
Slope5              -1.015e-02 -1.324e-01  1.120e-01     6749 0.859
Slope6              -1.183e-01 -3.516e-01  1.149e-01     5293 0.316
Slope7               1.057e-01 -6.565e-02  2.675e-01     6110 0.204
Slope8              -3.311e-01 -8.462e-01  1.715e-01     5824 0.199
```

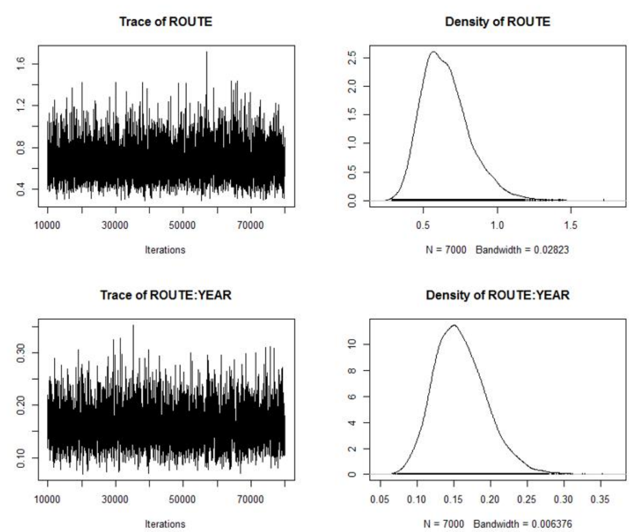
Appendix Dxxiv. Traceplots for Wood thrush populations in 20 m buffer



Appendix Dxxv. Traceplots for Wood thrush populations in 20 m buffer

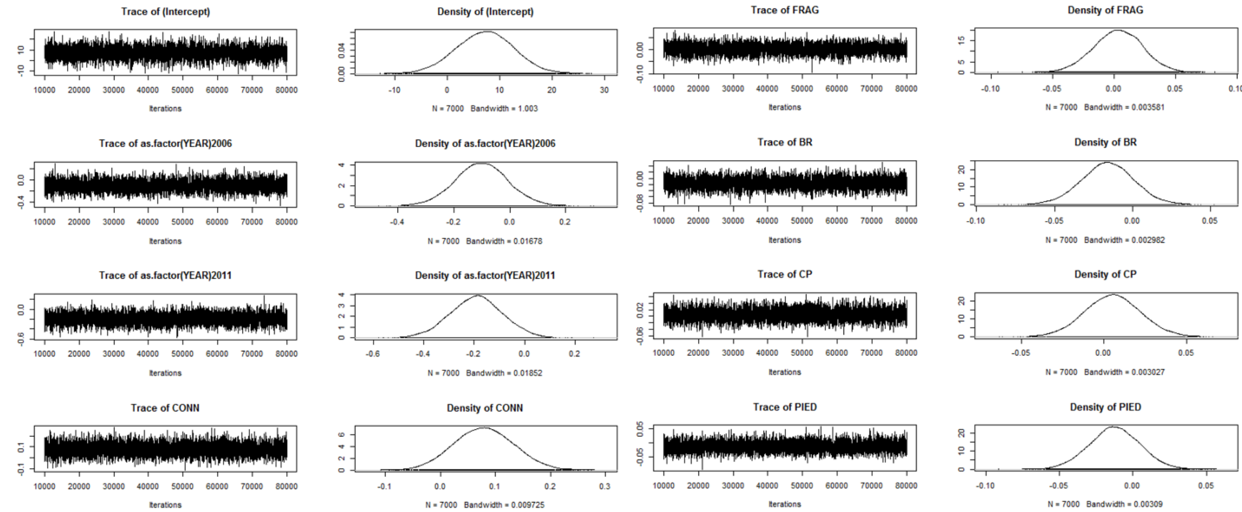


Appendix Dxxvi. Traceplots for Wood thrush populations in 20 m buffer and Model Summary Results

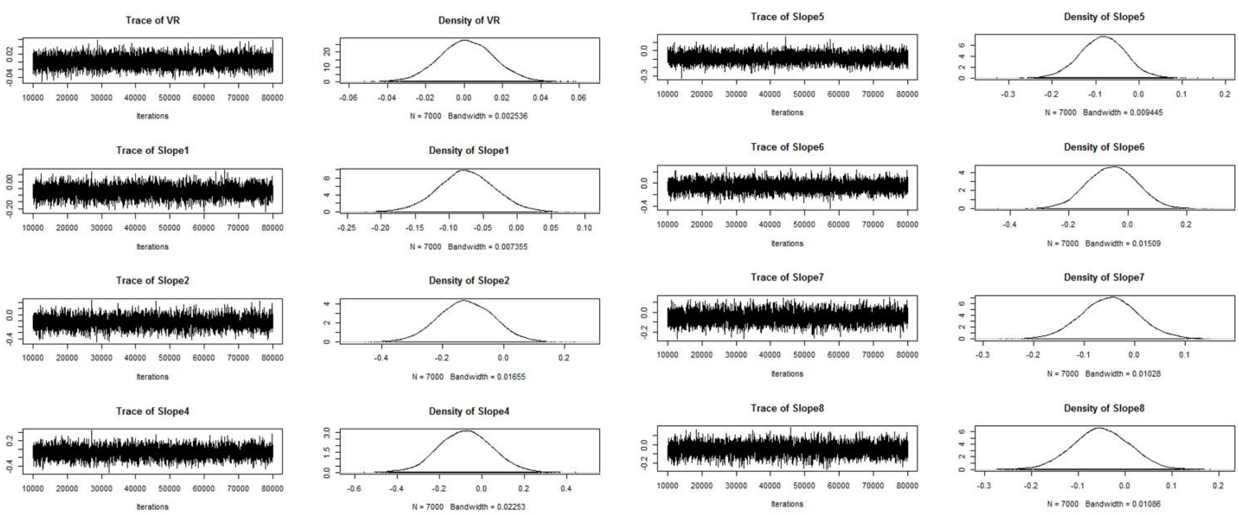



```
 Iterations = 10001:79991
 Thinning interval  = 10
 Sample size  = 7000

 DIC: 1111.307

 G-structure:  ~ROUTE

      post.mean l-95% CI u-95% CI eff.samp
ROUTE    0.6529   0.3709   0.9844     4206

 R-structure:  ~ROUTE:YEAR

           post.mean l-95% CI u-95% CI eff.samp
ROUTE:YEAR    0.1595  0.09191     0.23     3313

 Location effects: WTCOUNT ~ as.factor(YEAR) + CONN + FRAG + BR + CP + PIED + VR + Slope1 + Slope3 + Slope4 + Slope5 + Slope6 +
 Slope7 + Slope8

                    post.mean   l-95% CI   u-95% CI eff.samp  pMCMC
(Intercept)        -1.745e+00 -1.073e+01  7.789e+00     6681 0.7026
as.factor(YEAR)2006 -1.535e-01 -3.477e-01  2.135e-02     5980 0.0991 .
as.factor(YEAR)2011 -5.714e-01 -7.658e-01 -3.486e-01     6167 <1e-04 ***
CONN               -2.764e-02 -1.420e-01  7.975e-02     6192 0.6160
FRAG               -5.210e-03 -4.470e-02  3.443e-02     6445 0.7974
BR                 -3.402e-04 -2.938e-02  2.938e-02     7000 0.9866
CP                  8.988e-03 -2.260e-02  4.008e-02     7000 0.5671
PIED                1.024e-03 -2.996e-02  3.013e-02     7000 0.9320
VR                  1.421e-03 -2.389e-02  2.674e-02     6688 0.9040
Slope1              6.231e-02 -5.901e-02  1.802e-01     6490 0.3017
Slope3              7.727e-02 -8.787e-02  2.582e-01     6341 0.3771
Slope4             -8.349e-04 -9.825e-02  9.970e-02     6076 0.9883
Slope5              1.047e-01 -9.005e-02  2.812e-01     6566 0.2594
Slope6             -8.034e-03 -1.066e-01  8.940e-02     6144 0.8657
Slope7              1.030e-01 -6.473e-02  2.561e-01     6592 0.2023
Slope8             -1.441e-01 -4.657e-01  1.794e-01     6433 0.3706
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
```

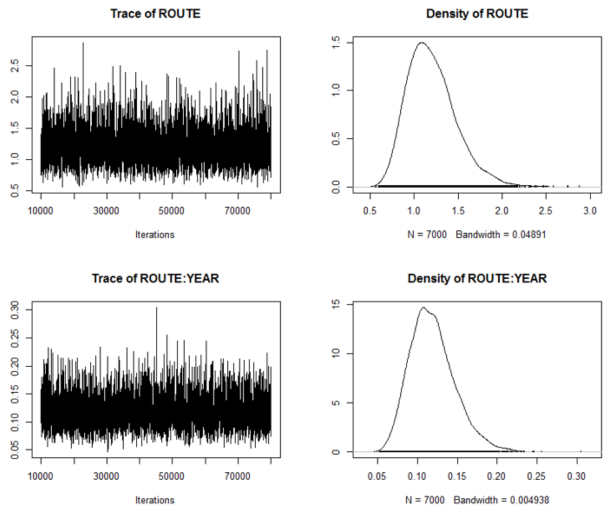
Appendix Dxxvii. Traceplots for Ovenbird populations in 25 m buffer



Appendix Dxxviii. Traceplots for Ovenbird populations in 25 m buffer

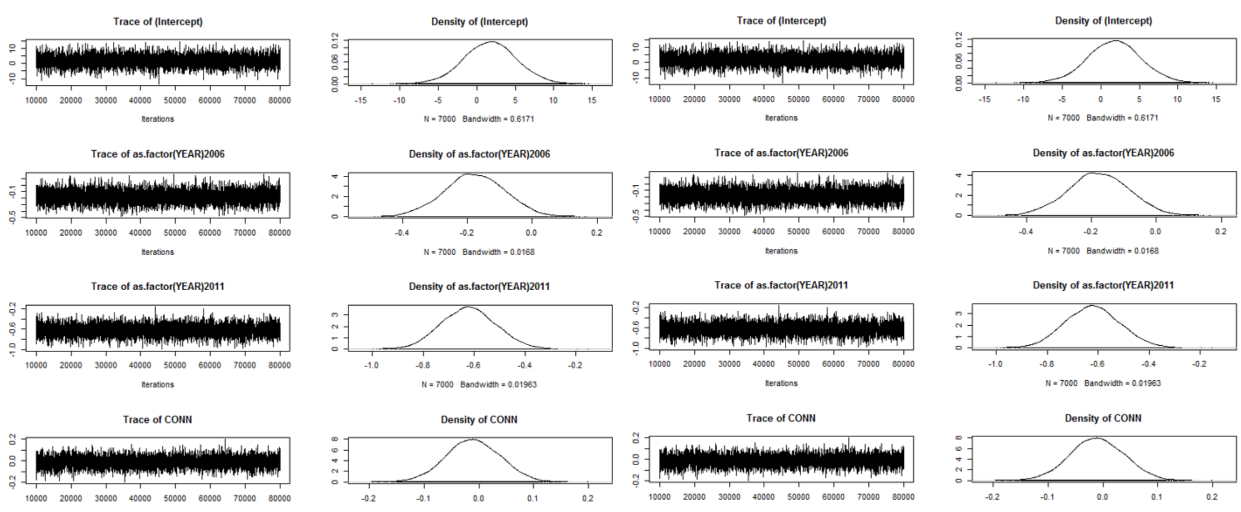


Appendix Dxxix. Traceplots for Ovenbird populations in 25 m buffer and Model Summary Results

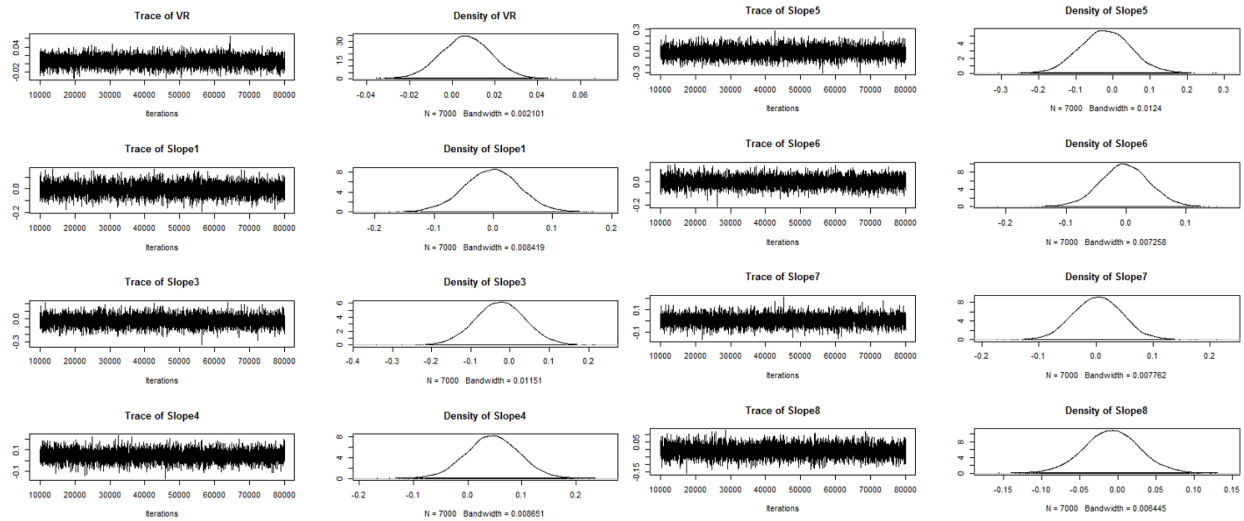



```
 Iterations = 10001:79991
 Thinning interval  = 10
 Sample size  = 7000

 DIC: 1086.499

 G-structure:  ~ROUTE

      post.mean l-95% CI u-95% CI eff.samp
ROUTE       1.2   0.7074    1.766     2885

 R-structure:  ~ROUTE:YEAR

           post.mean l-95% CI u-95% CI eff.samp
ROUTE:YEAR    0.1181  0.06857   0.1768     3113

 Location effects: OVCOUNT ~ as.factor(YEAR) + CONN + FRAG + BR + CP + PIED + VR + Slope1 + Slope2 + Slope4 + Slope5 + Slope6 +
 Slope7 + Slope8

                     post.mean  l-95% CI  u-95% CI eff.samp  pMCMC
(Intercept)           7.467147 -3.352594 18.429952     2598 0.1683
as.factor(YEAR)2006 -0.098856 -0.288667  0.076635     4716 0.2911
as.factor(YEAR)2011 -0.189177 -0.393085  0.008067     5450 0.0640 .
CONN                 0.079169 -0.020022  0.191110     5948 0.1377
FRAG                 0.002880 -0.036360  0.041532     6670 0.8817
BR                  -0.015644 -0.048011  0.016830     7000 0.3406
CP                   0.005505 -0.026654  0.039579     7000 0.7349
PIED                -0.012341 -0.045310  0.022218     6668 0.4549
VR                   0.001621 -0.027697  0.027992     7000 0.9180
Slope1              -0.076156 -0.166496  0.000668     3589 0.0706 .
Slope2              -0.128898 -0.307559  0.056161     2003 0.1697
Slope4              -0.081738 -0.343509  0.151298     2924 0.5206
Slope5              -0.083677 -0.183992  0.021179     5386 0.1100
Slope6              -0.056617 -0.228183  0.105737     2874 0.4940
Slope7              -0.051569 -0.162684  0.062546     2624 0.3651
Slope8              -0.052980 -0.167392  0.072660     2826 0.3923
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
```

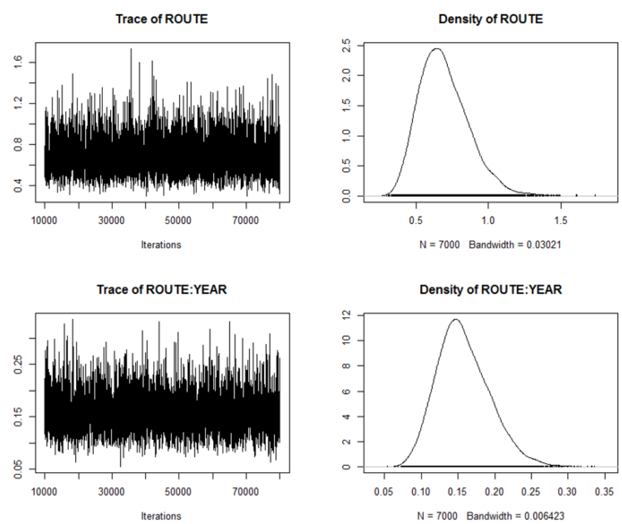
Appendix Dxxx. Traceplots for Wood thrush populations in 25 m buffer



Appendix Dxxxi. Traceplots for Wood thrush populations in 25 m buffer



Appendix Dxxxii. Traceplots for Wood thrush populations in 25 m buffer and Model Summary Results




```
 Iterations = 10001:79991
 Thinning interval  = 10
 Sample size  = 7000

 DIC: 1110.907

 G-structure:  ~ROUTE

      post.mean l-95% CI u-95% CI eff.samp
ROUTE    0.6946    0.397    1.038     4315

 R-structure:  ~ROUTE:YEAR

           post.mean l-95% CI u-95% CI eff.samp
ROUTE:YEAR    0.1579   0.0934   0.2315     3319

 Location effects: WTCOUNT ~ as.factor(YEAR) + CONN + FRAG + BR + CP + PIED + VR + Slope1 + Slope3 + Slope4 + Slope5 + Slope6 +
 Slope7 + Slope8

                     post.mean   l-95% CI   u-95% CI eff.samp  pMCMC
(Intercept)          1.7691924 -4.8987496  8.9403396     7000 0.6126
as.factor(YEAR)2006 -0.1772889 -0.3634711 -0.0018138     6416 0.0517 .
as.factor(YEAR)2011 -0.6214201 -0.8222296 -0.3998803     6842 <1e-04 ***
CONN                -0.0102792 -0.1069249  0.0856741     6150 0.8251
FRAG                 0.0111695 -0.0230357  0.0460905     6694 0.5320
BR                   0.0050923 -0.0217190  0.0309625     7000 0.6869
CP                   0.0051793 -0.0208709  0.0322217     7000 0.6926
PIED                -0.0008958 -0.0290092  0.0245130     7000 0.9514
VR                   0.0063306 -0.0178092  0.0277458     7000 0.5749
Slope1              -0.0027300 -0.0939596  0.0903029     7000 0.9651
Slope3              -0.0249122 -0.1504207  0.1016697     6758 0.6926
Slope4               0.0457099 -0.0479543  0.1427078     6762 0.3443
Slope5              -0.0204382 -0.1526128  0.1190352     7000 0.7680
Slope6              -0.0018475 -0.0838892  0.0767316     6750 0.9549
Slope7               0.0033873 -0.0844965  0.0847940     7000 0.9363
Slope8              -0.0084005 -0.0794672  0.0654812     7000 0.8166
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
```

Appendix E

THE IMPACT OF MANAGEMENT POLICIES ON FRAGMENTATION AND CONNECTIVITY

Abstract

Globally, management of ecological habitats with the aim of conserving biodiversity and promoting sustainable use of natural ecosystems remains a crucial issue. Ongoing research identifies processes and policies required for sustainable ecosystem management that include conservation and restoration efforts. However, there is much skepticism about ecosystem management as well as questions about the feasibility of proposed management policies. In this study, we assess management approaches and effects of human population density on natural ecosystems as major themes and identify the effectiveness of riparian buffers as corridors to aid in biological species conservation. We also examine the probability of adoption of a riparian buffer policy in Virginia. Our results show that densely populated counties in Virginia generally have smaller ecological patch sizes compared to sparsely populated counties. Also, there are very high percentages of ecological patches in Virginia that are currently unmanaged by either the federal government, the state, local or private bodies. Riparian buffers can serve as corridors to connect ecological patches in Virginia and landowners and farmers are willing to adopt a riparian buffer policy if they are given financial incentives.

Introduction

The habitat of biological species refers to areas that provide resources such as food, water and cover, and related conditions necessary to produce occupancy for those species (Hall et al., 1997). When a habitat is fragmented, smaller, isolated patches of the habitat separated by a matrix of other habitats, rather than the previous large expanse of the uniform patch, are formed (Wilcove et al., 1986).

Patch size, *isolation*, and *edge effects* are three main effects of habitat fragmentation. By reducing habitat patch sizes, movement barriers are created, ultimately causing a reduction in species population sizes (Bender et al., 1998). Isolation effects include a reduction in access to resources and can lead to inbreeding, causing genetic abnormalities and weaknesses (Young et al., 1996). Fragmentation can also result increase edge effects, leading to changes in abiotic conditions such as increased sunlight penetration and wind speeds, as well as in biotic conditions such as increased predation risk and invasions (Hennings and Soll, 2010).

Large ecological patches have greater benefits than the same total area of smaller patches (Wintle et al., 2019). Some of these benefits stem from species-area relationships that increase species richness in a given habitat as habitat size increases (Wintle et al., 2019). There is a greater variety of habitats in larger patches as compared to smaller patches as well as greater protection from disturbance from adjacent areas. Larger ecological patches provide important ecosystem services, including nutrient and pollutant filtration, prevention of soil erosion and carbon sequestration (Zari, 2015). As ecological patches become smaller, their ability to provide important ecosystem services are significantly reduced, as are the lower numbers of species they are able to support.

For instance, smaller forest areas formed as a result of forest fragmentation are less resilient to severe weather conditions and disease outbreaks (Allan et al., 2003, Opdam and Wascher, 2004). Sensitive interior species that require deep cover within interiors of continuous habitats are adversely affected as their ability to survive and reproduce is reduced. Most interior forest birds such as the Wood Thrush and

Ovenbird experience increases in nest and adult predation by edge-dwelling species such as the Brown-Headed Cowbird and raccoons, as forest fragmentation leading to the expansion of edge areas, increases vulnerability (Sisk and Battin, 2002). Design of efficient and effective forest management policies should consequently consider both the total area of forest habitats conserved as well as its spatial configuration (Albers and Bu, 2009; Lewis and Alig, 2009; Albers et al., 2010; Sims 2013).

Although earlier research studies considered costs and benefits of differing habitat patterns and the expected impacts of future policies on habitat fragmentation (Lewis and Plantinga, 2007; Horan et al.,2008; Lewis et al., 2009; Ando and Shah, 2010; Lewis, 2010; Lewis et al., 2011), a rigorous evaluation of consequent impacts of management policies has not been conducted. Other studies have used quasi-experimental methods to evaluate impacts of conservation policies on losses of habitat or of species (Ferraro et al., 2007, Costello et al. 2008, Andam et al., 2010; Sims, 2010), but very few studies have used those methods to evaluate impacts of management policies on forest habitat fragmentation, forest connectivity and threats posed to biodiversity.

Restoration and retention of forest ecosystems have recently become a global concern due to significant losses within certain taxonomic groups. For instance, 13% of bird species have been lost globally due to changes in forest ecosystems (Hoffmann et al., 2010). As a result of current global trends, one of the most important emerging questions is how to manage ecological areas for economic productivity while conserving biodiversity and improving other important ecosystem services (Duncker et al., 2012). In order to do this, it is important to anticipate long-term effects of management policies and strategies on the status and dynamics of ecosystems (Duncker et al., 2012). Ecosystem management is rarely successful when only single species are considered; more effective management policies act through sets of operations aimed at multiple variables and species (Duncker et al., 2012).

Habitat fragmentation leading to less interaction between species in different patches is a contributing factor to the pattern of decreasing species populations. Due to continuing trends of fragmentation, it has become important to maintain and restore structural heterogeneity by establishing structures that ensure habitat connectivity (Brockerhoff et al., 2008). Habitat connectivity is important to discourage inbreeding among species, and ultimately, to ensure healthy species populations (Olson and Burnett, 2013). Construction of multiple connective pathways ensures that species are able to move when site-specific factors, such as availability of food or increased predation, change. Thus, increasing numbers of pathways for movement enhances persistence of species population (Olson and Burnett, 2013).

As ecosystems such as forests become increasingly fragmented due to human activities such as road construction, agriculture and other developments, corridors of trees can provide vital pathways for contributing to species diversity in ecosystems (Naiman et al., 2003). Riparian buffers can serve as corridors, functioning as connections between terrestrial ecosystems (Naiman et al., 2003). Riparian buffers as pathways for ecosystem connectivity have the potential to not only link ecosystems but serve as frameworks for understanding the organization and diversity of ecosystems as they comprise sharp environmental gradients and various species communities (Naiman et al., 2003).

This study identifies ways to make existing ecosystem management policies more effective by incorporating knowledge of impacts of urbanization on ecological patches in Virginia, and by identifying ways to improve connectivity between ecological patches. By examining current densities of ecological patches in both densely and in sparsely populated counties in Virginia, this study highlights how human populations influence the ecological integrity of an area. Patch metrics in specific counties in Virginia are examined to identify the relationship between population density ecological patches. Conditions in conservation lands in Virginia that are prioritized and are therefore managed by the state, federal government, an identified private or local body, are more stable over time. This study aims to identify how many ecological patches in Virginia are classified as conservation lands and are therefore, intentionally managed. Also, use of riparian buffers as effective corridors that connect ecological patches are examined, as is their feasibility in Virginia. To do this, hypothetical riparian buffers using geospatial skills are created around large waterbodies. Survey questionnaires are sent out to identify the perceptions of farmers and landowners regarding the riparian buffers. In this study, we specifically answer the following questions:

1. What is the relationship between human population density and ecological patch integrity in Virginia?
2. How many ecological patches in Virginia are being protected?
3. How can riparian buffers improve patch connectivity in Virginia?
4. How feasible will the establishment of riparian buffers in Virginia be?

Method

Data

Data collection for this objective entails a combination of geospatial data, literature review and survey questionnaire responses from stakeholders. Survey questionnaires were generated and distributed using Qualtrics software (version CX). Hydrologic data showing Virginia's waterbodies were obtained from the United States Geological Survey. Information on human population density of counties in Virginia was obtained from the United States Census Bureau.

In addition, two products from Virginia's Department of Conservation and Recreation (DCR) were examined (Figure 1). The first product from DCR is the Virginia Natural Landscape Assessment (VaNLA) project. Land cover satellite imagery is used to identify natural land patches that have at least one hundred acres of interior cover. The project assigned an ecological integrity value based on a patch's ability to produce ecosystem services such as clean air and water, its capacity to support biological species, or its ability to provide food and other materials such as timber. Depending on the overall score of a patch, it is categorized as *General*, *Moderate*, *High*, *Very High* or *Outstanding*. Generally, higher scores are attributed to patches that are large, and provide the most biologically diverse areas.

The second product from the DCR shows *managed conservation lands* in Virginia. Collection of GIS data for this map by the DCR began in 1998, as collected by state and federal land management agencies and localities, as well as local and private landowners. This data set is continually updated with the latest information (in this case, November, 2018). Management of conserved lands is categorized as Federal, State, Private or Local.

Methodology

In ArcGIS, the population density for every county in Virginia was calculated using Zonal Statistics tools. The five counties with the lowest population densities and the five counties with the highest population densities were identified. The metrics of ecological patches in these counties were examined and compared to identify the relationship of human population density to ecological patch quality in Virginia. Patch metrics such as mean patch size, smallest patch size, largest patch size and patch density are calculated.

To identify how many ecological patches in Virginia are protected, we examined the two products from Virginia's DCR showing the number of ecological patches in the state, and conserved lands, using Intersection tools in ArcGIS. For each ecological patch category (such as *Outstanding* or *General*), the numbers managed by either State, Federal, Local or Private bodies were calculated.

Using hydrologic data from the United States Geological Survey, large waterbodies in Virginia with a total area of 10 square miles or more, were selected. Then, we created 100ft buffers around these waterbodies using the Buffer tool in ArcGIS. These buffers were examined to verify if they could serve as corridors connecting General Ecological Patches to each other using the Intersect and Zonal Statistic tools. The General Ecological Patches are chosen for the riparian buffer analysis because they are generally smaller in size and are more likely to have small species population sizes at risk of extinction.

We prepared a survey questionnaire designed to evaluate stakeholder perceptions of riparian buffers. The survey questionnaire was reviewed by Virginia Tech's Human Research Protection Program to ensure that questions asked meet the Institutional Review Board (IRB) regulations. Stakeholders who received this questionnaire included farmers and landowners in Virginia. The questionnaire was distributed through social media links and extension services listservs. Then, an analysis based upon existing literature was applied to assess the Riparian Buffer policy in Virginia to identify reasons for their limitations and successes.

Results

Figure 2 depicts the population density of Virginia on a county basis. The five counties with the highest population densities in Virginia are Chesterfield, Prince William, Henrico, Fairfax and Arlington, with Arlington having the highest population density. The county with the lowest population density in Virginia is Highland County followed by Bath, Northampton, Craig and Bland in increasing order.

Results show that the average size of the smallest ecological patch in counties with the lowest population density in Virginia was higher than that of the counties with the highest population densities (Table 1). While the average smallest patch size among the five counties with high population densities is approximately 7,405m^2, the average smallest patch size for counties with low population densities is about double that size ranging, 14,191m^2. On average, the largest ecological patch in the 5 counties with high population densities is about 19,008,157m^2 compared to an average of 108,419,062m^2 in sparsely populated counties. When the mean size of ecological patches in each of the 10 counties was calculated, the average mean size of patches in the five counties with the highest population density was 1,193,635m^2, compared to 9,926,542m^2 among the counties with the lowest population densities in Virginia.

When different categories of ecological patches were compared among the 10 counties, Chesterfield, the county with the fifth highest population density, recorded the largest number of Outstanding

Ecological Patches, followed by Prince William County, another county with very high population density. The highest number of General Ecological Patches was recorded in Craig County, one of the counties in Virginia with the lowest population density (Table 1).

Approximately 63% of Outstanding Ecological Patches are situated on lands that are not managed or conserved (Figure 3). 33% of Outstanding Ecological Patches are on federally owned and managed lands while 5% are on state lands. Local lands have the smallest percentage of Outstanding Ecological Patches (Figure 3).

The highest percentage of Very High Ecological Patches are found in lands that are not managed or conserved by either the federal, state, local or private bodies (Figure 4). 23% of Very High ecological patches are managed by the federal government while 5.5% are managed by the state. Privately managed lands have a higher percentage (1.4%) of Very High Ecological Patches compared to Outstanding Ecological Patches.

Figure 5 shows the proportion of High Ecological Patches managed by either the federal government, the state, local or private bodies. While the percentage of High Ecological Patches managed by the federal government is lower at 8% compared to 33% and 23% for Outstanding and Very High Ecological Patches respectively, the percentage unmanaged or found on lands that are not conserved, is higher at 89%. This trend continues for the Moderate and General Ecological Patches with the percentage of General Ecological Patches found on lands that are not conserved reaching approximately 95% (Figures 6 and 7).

A total of 376 General Ecological Patches become connected as a result of the hypothetical 100ft buffer around waterbodies greater than 10sqm in Virginia (Figure 8). 12 groups of previously independent General Ecological Patches are formed as a result of buffers thus created. The largest group of General

Ecological Patches consist of 109 patches formed around the Staunton River and the John Kerr Reservoir.

100 Virginia residents responded to the survey questionnaire. 97% of the respondents were landowners, with the remaining being farmers. About 75% of the respondents ranked riparian buffers very valuable attributing a value of 5 on a scale of 1 to 5 to riparian buffers (Figure 9). Figure 9 shows that approximately 16% of respondents chose 4 as the value of riparian buffers, and 3.5% of the respondents put the value of riparian buffers at 1. While 60% of the respondents were willing to establish riparian buffers on their lands, about 6% were unwilling and 34% were unsure and therefore opted for the 'Maybe' option (Figure 10). About 63% of the respondents wanted tax reductions as compensations for establishing riparian buffers on their lands. None of the respondents wanted subsidized loans as compensation. Approximately 36% of the respondents sought money for establishing riparian buffers on their lands, while only 1% of the respondents preferred fertilizer and vaccines in exchange for establishing riparian buffers (Figure 11).

Discussion

Consistent with our initial hypothesis, counties with high human population densities generally have fewer large ecological patches compared to counties that have lower population densities. The difference is evident in both the average small patch and large patch sizes of ecological patches in these two categories of counties. Because increased human population in an area leads to fragmentation of that area (Knight and Fox, 2000), it can be difficult to find large ecological patches in counties that have high population densities. With the exception of a few areas, habitat fragmentation leading to the creation of small patches in an area occurs simultaneously with increasing human disturbances in that area (Knight and Fox, 2000; Soulé et al., 1992; Laurance, 1997; Smith, 1997). Studies that have included increasing human disturbances as a factor influencing habitat fragmentation have demonstrated the significance of human population (Knight and Fox, 2000; Abensperg-Traun et al., 1996; Dunstan and Fox

1996). It therefore is not surprising that our study shows that counties with high population densities have smaller ecological patch size on the average, compared to those counties that have low population densities.

Disturbance of ecological areas leading to habitat fragmentation due to human interference can be difficult to understand. In certain cases, human disturbance in the form of mining and timber logging can lead to fragmentation (Knight and Fox, 2000). This anthropogenic cause of fragmentation can occur irrespective of the actual human population density of the area. This study is specific to human population density and therefore triggers discussions on how development such as constructions of roads, residential homes and buildings, can lead to fragmentation. It is important for decision makers and city planners to identify ecological patches in an area and make efforts to protect those areas when planning urban developments in order to conserve biodiversity.

Our study highlights the importance of identifying and including information on the ecological integrity of urban centers when planning urban developments in the form of construction of buildings. It is interesting that among the 10 counties assessed, the highest number of Outstanding Ecological Patches are found in a county with very high population density. This means that although low population areas have more patches, most of them are of low quality. As the pace and pattern of urban growth intensifies, finding ways to sustain ecological patches in urban regions will determine the future of ecosystems (Alberti, 2010).

Interactions between humans, population growth and ecosystem function are controlled by multiple agents and mechanisms that operate at a variety of temporal and spatial scales (Alberti, 2010). Policies aimed at controlling such interactions should be targeted especially at areas with high population density. Results of this study showing ecological patches with high integrity values in highly populated counties provide basis for such policies. Policies such as imposition of higher taxes on residents of highly populated counties can serve as effective deterrents against activities that degrade ecosystem quality and also cover environmental costs required for maintaining the integrity of the ecological patches in those areas. Due to the high interaction between humans and the natural ecosystem, policies on recycling and improved energy use efficiency in these areas have the capacity to slow the rate of resource erosion (Zenher, 2012).

The debate over whether to conserve biodiversity only on the basis of its services for human or on their value for nature regardless of human use continues as biodiversity continues to decline globally (Tittensor et al., 2014; Marvier, 2014; Pearson, 2016). Pearson (2016) makes the argument for policies geared towards natural conservation to include utilitarian values in order to avoid the perception that nature conservation is at odds with human progress and that conservationists value nature above human needs. It is important that areas that have ecological value are managed for both biodiversity conservation and human use. The fact that Figures 2, 3, 4, 5, and 6 show that many of the areas with ecological patches in Virginia are in areas that are neither managed by the state, federal government, local or private bodies, is worrying.

Natural ecosystems function as the main climate regulators (The Economics of Ecosystems and Biodiversity, 2010). Ecosystems currently absorb about 50% of all anthropogenic CO_2 emissions (Mumba et al., 2017). It is worrying that ecosystem absorptive capacity is declining by about 1% per decade and is likely to decline more rapidly due to climate change and human impacts (World Bank, 2009). In order to mitigate the negative impacts of ecosystem degradation, it is important that the state, federal government, local and private management bodies increase their conservation efforts. Policy makers in Virginia should use the results of this study showing such high percentages of unmanaged ecological patches as a basis to increase conservation efforts and stipulate regulations for the effective management of ecological patches.

Although General Ecological Patches are smaller in size, small patches represent an increasingly large component of the remaining habitat in many ecosystems (Tulloch et al., 2015). Our results showing high numbers of unmanaged General Ecological Patches in Virginia raises concerns about how small patches that play significant ecosystem roles remain neglected. This result is consistent with studies that show that small patches receive very little conservation attention compared to larger ones (Biber et al., 2015; Gamfeldt et al., 2013). Conservation policies should be targeted at the management of small ecological patches, prompting these patches to be put under the supervision of a managing body.

Riparian buffers effectively serve as corridors to connect ecological patches, ensuring interaction between species in different patches. Such interactions are important in maintaining growth and expansion of species within a given landscape (Peters et al., 2006). Riparian buffers are effective also because of their role in hydrological ecosystems as they decrease pollution, control erosion, and serve as habitats for wildlife. Especially in regions such as Virginia that have high agricultural activities, pollution from fertilizers and animal waste threaten aquatic organisms as these pollutants reduce water quality (Tomer et al., 2005). Agriculture continues to be the largest private industry in Virginia employing more than 50,000 workers (Rephann, 2017). Increasing agricultural activity increases likelihood of pollution of waterbodies (Tomer et al., 2005). Thus, a riparian buffer policy aimed at connecting ecological patches will play multiple functions to support conservation of Virginia's biodiversity.

Our results illustrating effectiveness of riparian buffers as connectivity measures between ecological patches highlights the role of riparian buffers in conservation. With more than 300 General Ecological Patches connected as a result of 100ft buffers around only large waterbodies with a total area above 10 sqm, riparian buffers have the potential of not only serving as habitats for wildlife but improving the ecological integrity of already existing ecological patches. With more General Ecological Patches connected as a result of riparian buffers, mobility of species between the patches will be enhanced and interaction between different populations improved. In order to sustain ecosystem processes such as trophic and species interactions, species mobility must be a significant consideration (Massol et al., 2011). Increasing species mobility encourages species interactions that have cascading effects on food web dynamics and persistence of species populations (Hagen et al., 2012). Hence, a policy that ensures that landowners and farmers in Virginia establish riparian buffers should their lands include waterbodies, forms a significant step towards biodiversity conservation.

The feasibility of a riparian buffer policy in Virginia is examined in this study. With a prerequisite of the questionnaire being that respondents are either farmers or landowners, a 97% landownership response shows that landowners in Virginia are usually administratively in charge of activities on their land. According to Liu et al. (2018), landowners are more likely to adopt best management practices such as riparian buffers that benefit either themselves or the environment in the long run. This is because renters only operate on lands over a relatively short amount of time and therefore do not benefit from policies whose effects are only visible after a long term (Liu et al., 2018). With most of the respondents being landowners, a riparian buffer policy as well as other best management policies should be relatively easy to adopt if effective measures are proposed by policy makers.

The fact that 75% of the respondents highly valued riparian buffers shows the possibility of an effective riparian buffer policy in Virginia. However, there is room for more education if all landowners in Virginia will establish riparian buffers on their properties. Dutcher et al. (2004) showed that most conservation-minded landowners and farmers lacked access to understandable and reliable information. It is important for the state to provide educational programs and technical assistance for landowners and farmers in Virginia so that they will understand the importance of riparian buffers for conservation. The functional value of riparian buffers increases dependent on the vegetation used. Hence, it is important for extension agents to increase knowledge among farmers and landowners concerning riparian buffers in Virginia.

The need for educational programs and technical assistance for landowners and farmers in Virginia is evident in the reduction in percentage of respondents willing to establish buffers on their properties, compared to respondents who place high value on riparian buffers. Our results are consistent with prior studies that have shown stronger landowner support for the values of riparian buffer systems than for the actual practice of establishing riparian buffers (Hairston-Strang and Adams, 1997; Schrader, 1994). While 60% of the respondents are willing to establish riparian buffers on their properties, about 34% of the respondents remain undecided. Given that a higher percentage (75%) know about the value of riparian buffers but only 60% are willing to establish them, there is a need to consider mechanisms that might make it difficult for landowners and farmers in Virginia to establish riparian buffers.

Johnson (1996) argued that although landowners do not want to damage waterbodies, existing policies for managing riparian buffers on privately owned lands are not flexible enough to allow for innovation and site-specific circumstances. For instance, Virginia's riparian buffer management stipulates that only indigenous vegetation be used as vegetation. This might pose a problem for landowners concerned with aesthetics and those who will have to make an effort to purchase indigenous plants. In Maryland, a roundtable of experts reached the conclusion that costs of planting and maintaining riparian buffers, the lack of understanding of the value of riparian buffers, uncertainty among landowners and farmers about extension agency locations, use of stream lands for recreational purposes, aesthetic reasons and the fear of government control are among the reasons why landowners and farmers are unwilling to establish riparian buffers (Eastern Shore Tributary Strategy Teams, 1997). This calls for widespread education among landowners and farmers by extension agents on riparian buffer establishment, management and their contribution to conservation.

Prior studies have identified tax relief, compensation and cooperative agreements as preferred policy options for influencing riparian buffer establishments (Dutcher et al., 2004; Hairston, 1996; Johnston, 1996). Our results showing over 60% of respondents requiring tax reductions confirm the importance of tax reductions to landowners and farmers in Virginia. In Virginia, "highest and best use" of land, an attribute that increases its real estate value, assumes that the land is fully developed. With many landowners having no desire to develop their lands, they do not benefit from the bound-up value (McLaughlin, 2004). A conservation easement on the property in the form of riparian buffer establishment, means that they have chosen to forgo some of the land value by not developing it and this should yield tax benefits. With so many undeveloped areas on private properties, proper education on the value of riparian buffers in Virginia and the provision of financial incentives, have the potential of improving the quality of waterbodies and enhancing aquatic life, while simultaneously, providing benefits for landowners and farmers.

Financial incentives in the form of monetary compensation are important in the adoption of best management practices, as collaborated by the 35% of our respondents who chose to receive money for riparian buffer establishment (Welch and Marc-Aurele, 2001). Liu et al. (2018) showed that landowners or farmers who have more financial resources are more likely to adopt BMPs compared to those that are not wealthy or receive their income only through farming. For landowners to adopt BMPs, Läpple and Hennessy (2014) found that the lack of money needed to offset the cost of BMP establishments is one of the barriers to adoption. Hence, it is important for policy makers to provide landowners and farmers with incentives, possibly including financial support to purchase tree seedlings and pay for labor, if rate of adoption of best management policies such as the riparian buffer policy, is to be increased.

Conclusion

Human population density affects the ecological integrity of ecosystems in Virginia through fragmentation. Policy makers should identify effective policies that ensure urban development, as a result of increasing human population, does not affect ecosystem functioning. In highly populated areas with high numbers of patches with significant ecological integrity, the imposition of higher taxes on residents whose activities degrade environmental quality, recycling and energy use policies can help in biodiversity conservation.

Our study shows that management efforts to conserve ecological patches in Virginia should be increased. With most of the ecological patches currently unmanaged, there should be concern about their state. Our findings highlight effects of neglect of small ecological patches in Virginia and the need to focus efforts in their management.

Riparian buffers serve as effective mechanisms for ecosystem management and conservation by improving water quality in aquatic ecosystems and by serving as corridors that connect ecosystem patches and provide habitats for ecological species. Increasing education among farmers and landowners in Virginia will improve awareness and understanding of the importance of riparian buffers and the likelihood of adoption of riparian buffer policies. Tax reductions and monetary compensations can also encourage adoption of riparian buffer policies in Virginia.

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Figures and Tables

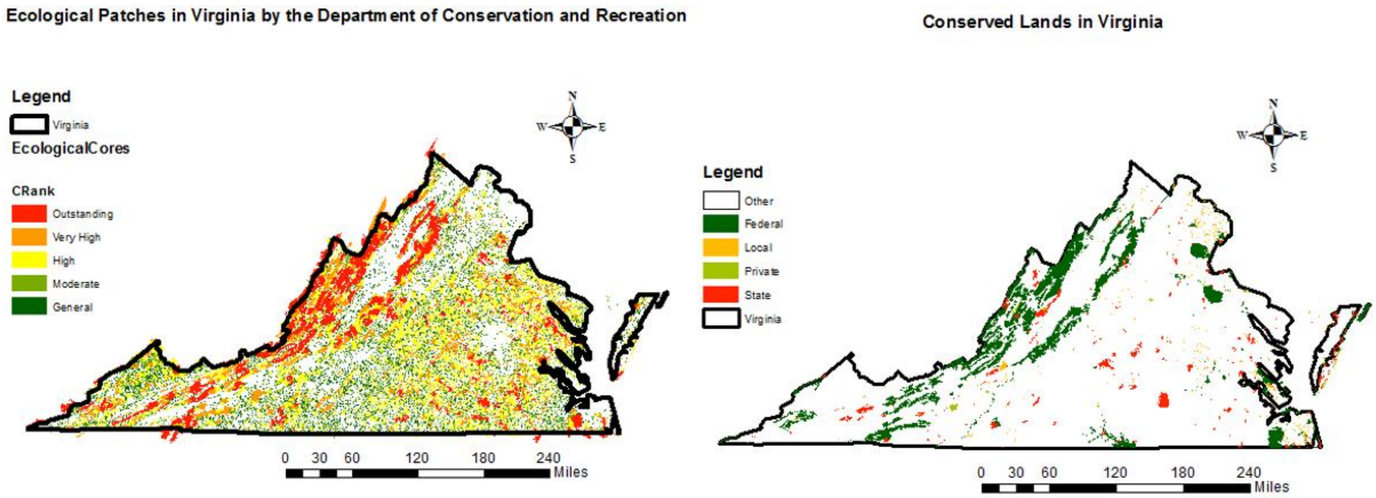

Figure 1. Products from the Virginia Department of Conservation and Recreation. The first map shows the ecological patches in Virginia and the right map shows the lands that are managed or conserved.

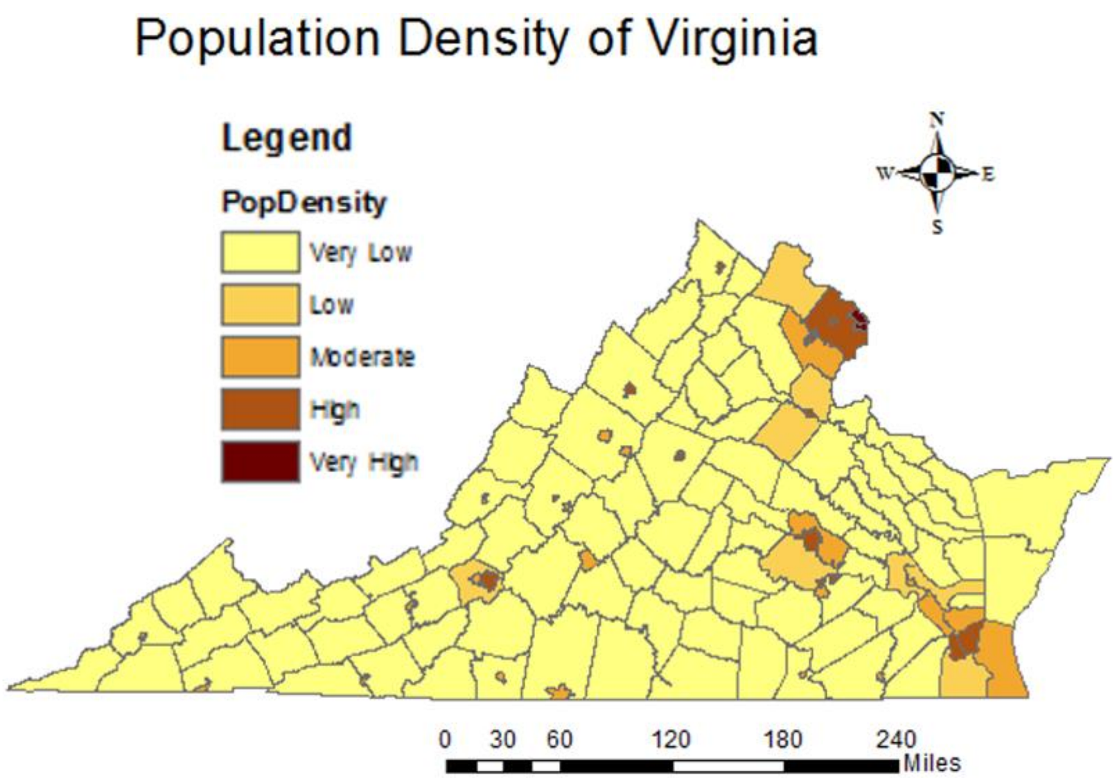

Figure 2. A population density map of Virginia. Population density is calculated by county.

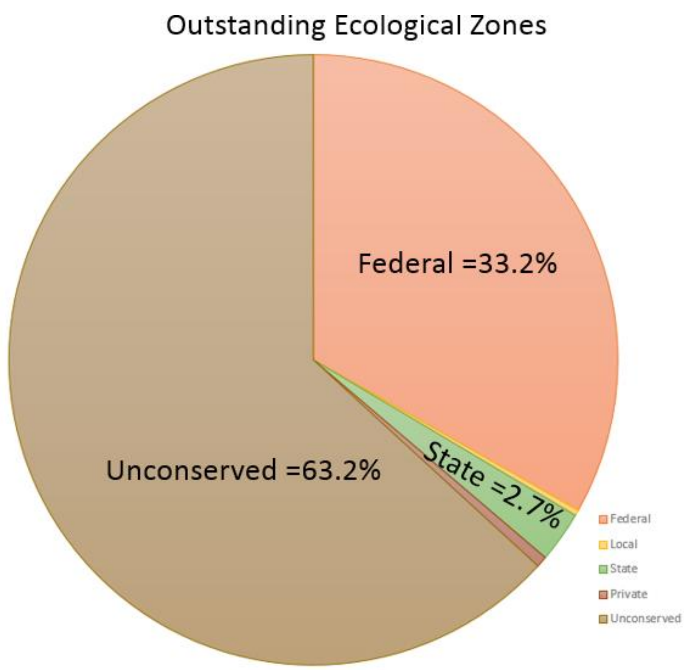

Figure 3. Conservation status of outstanding ecological patches in Virginia. Most of the Outstanding Ecological Patches are not on managed lands.

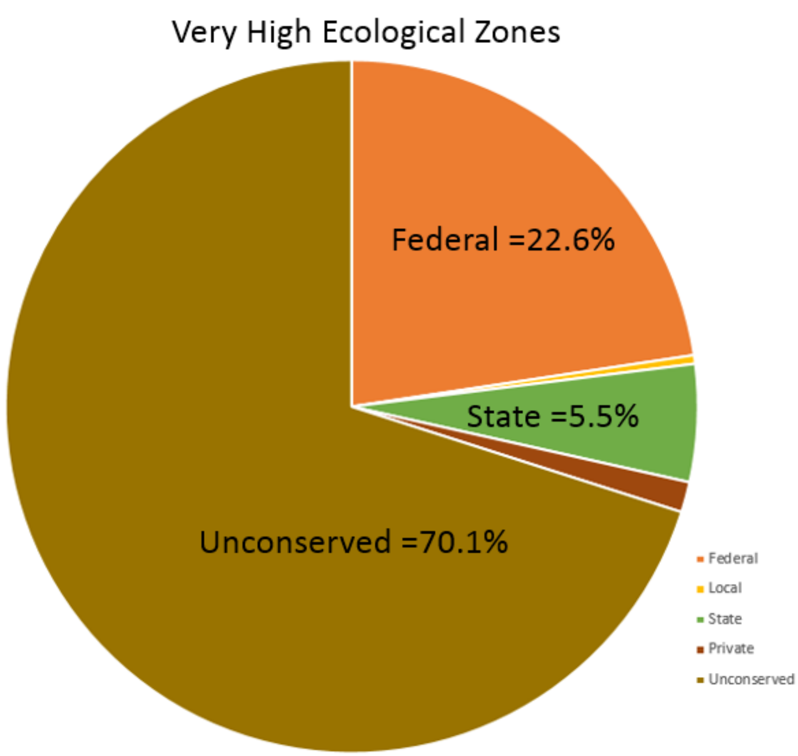

Figure 4. Conservation status of Very High ecological patches in Virginia. Over 70% of the Very High Ecological Patches are not on managed lands.

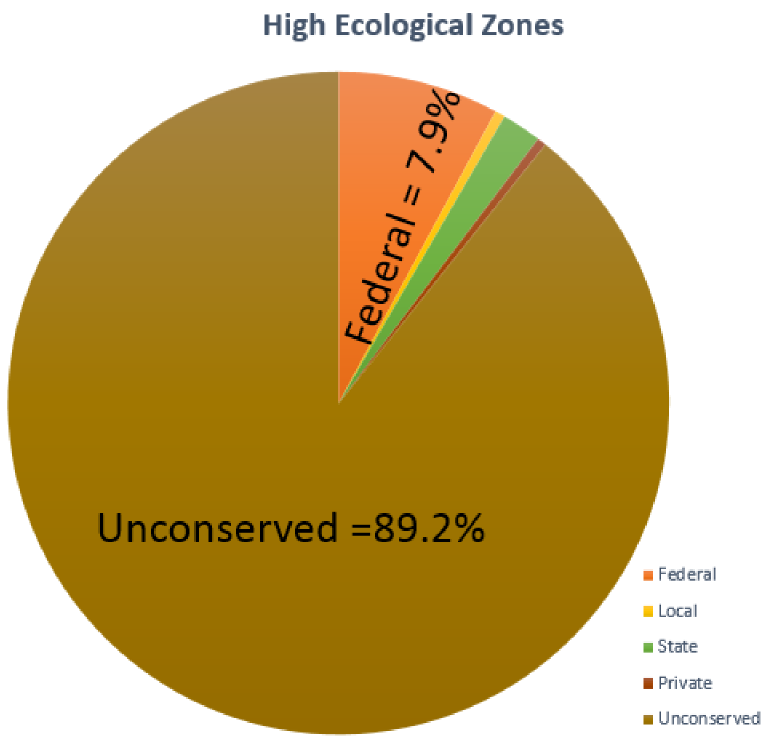

Figure 5. Conservation status of High ecological patches in Virginia. More than 89% of High Ecological Patches are not on managed lands.

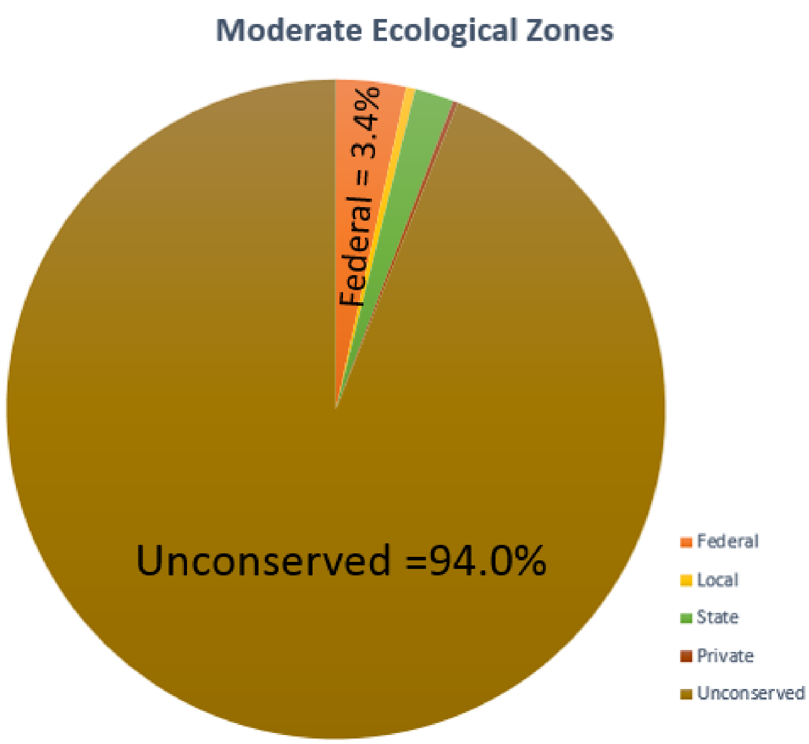

Figure 6. Conservation status of Moderate ecological patches in Virginia. 94% of Moderate Ecological Patches are not on managed lands.

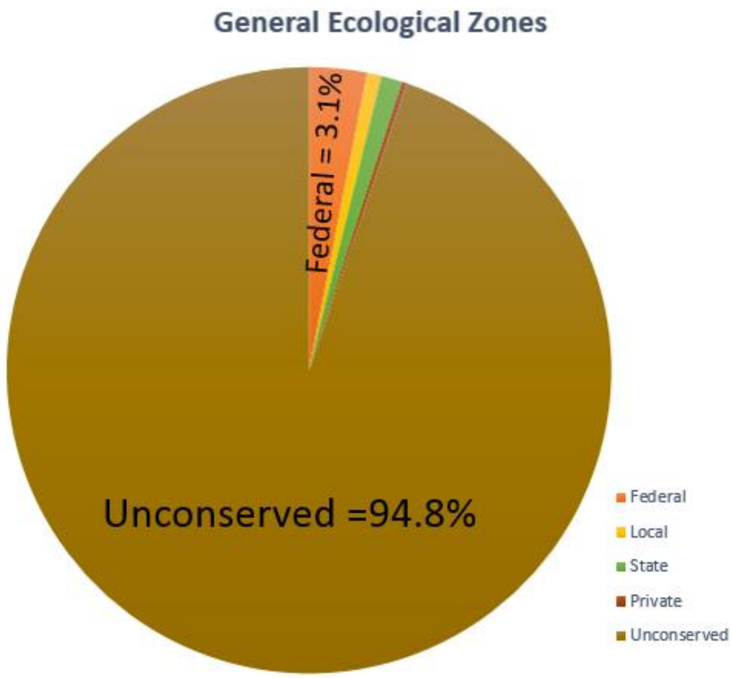

Figure 7. Conservation status of General ecological patches in Virginia. More than 94% of General Ecological Patches are not on managed lands.

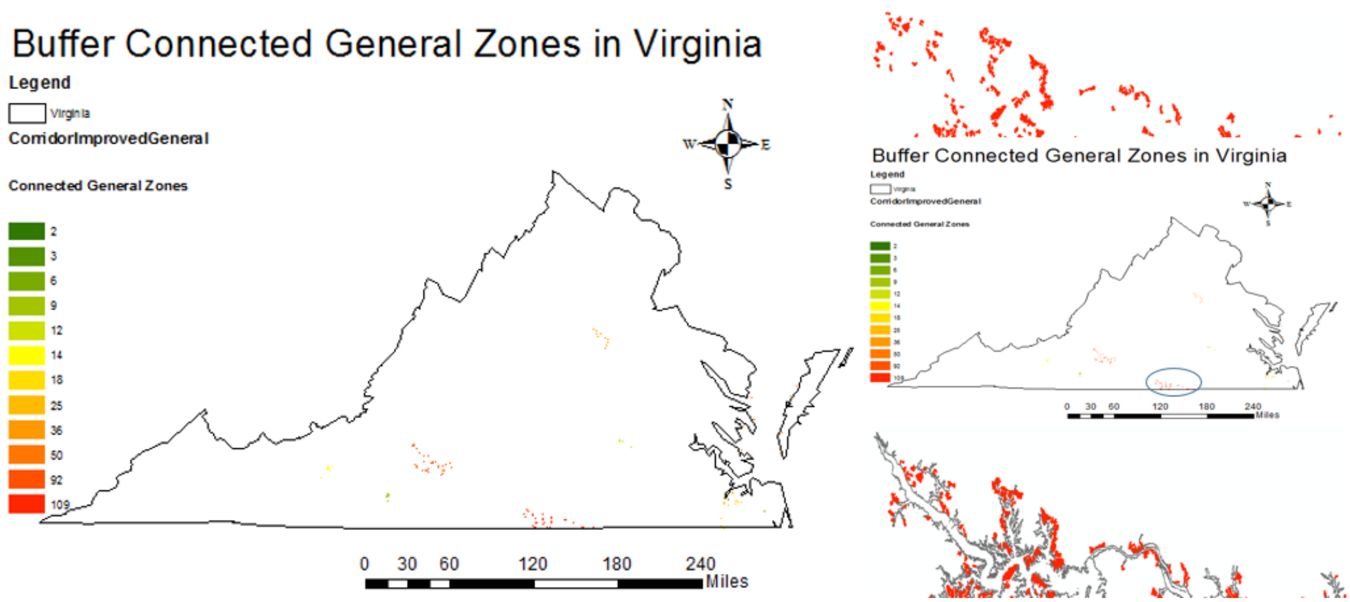

Figure 8. A map showing the General ecological patches connected as a result of the 100ft buffer. The indexed image on the right shows the largest cluster formed as a result of the 100ft buffer around the Staunton River below.

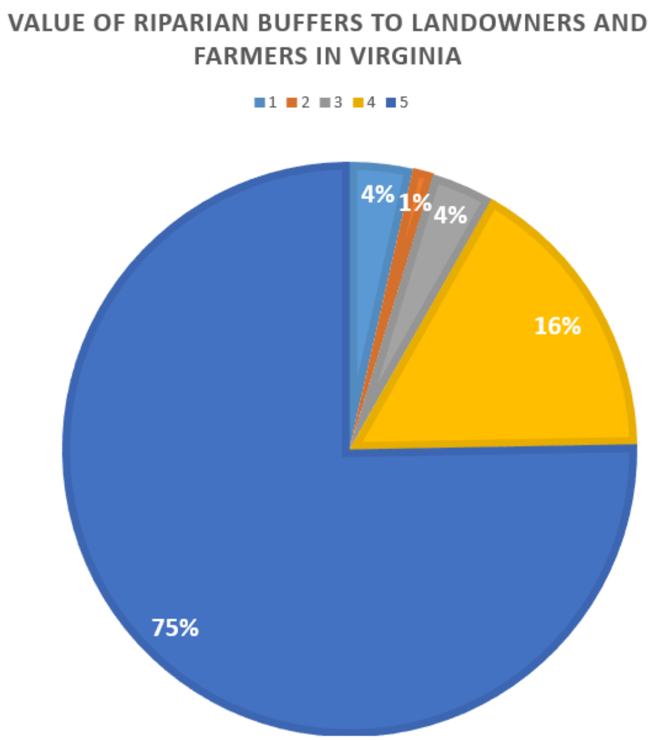

Figure 9. Farmers and Landowners in Virginia show how they value riparian buffers on a scale of 1 – 5 with 5 being the highest value. 75% think that riparian buffers are very valuable to the ecosystem.

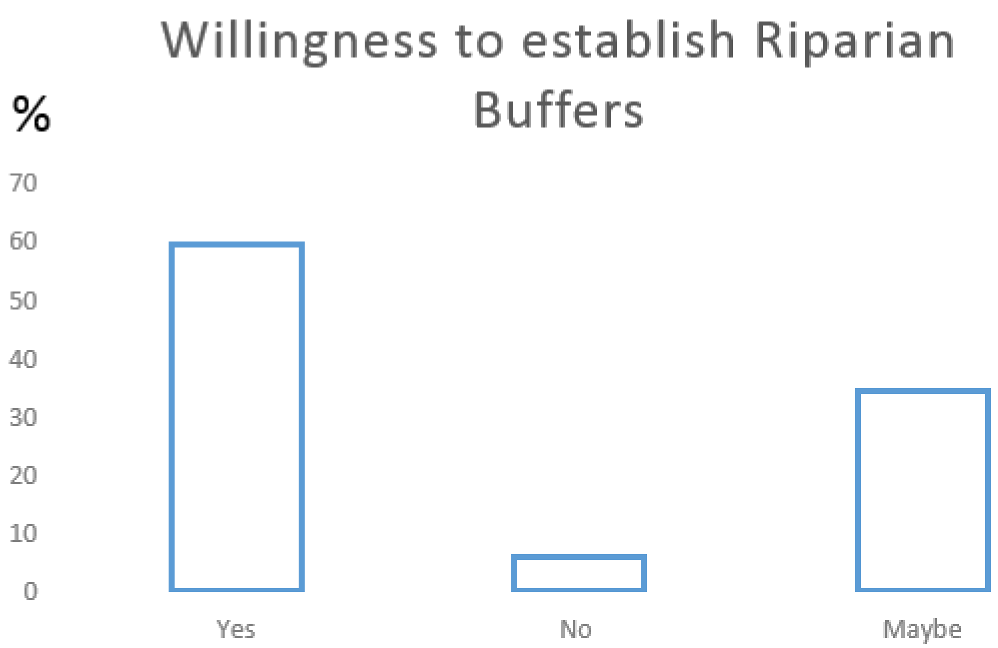

Figure 10. Willingness of farmers and landowners to establish riparian buffers on their property. Only a small percentage of respondents are sure that they do not want riparian buffers on their lands.

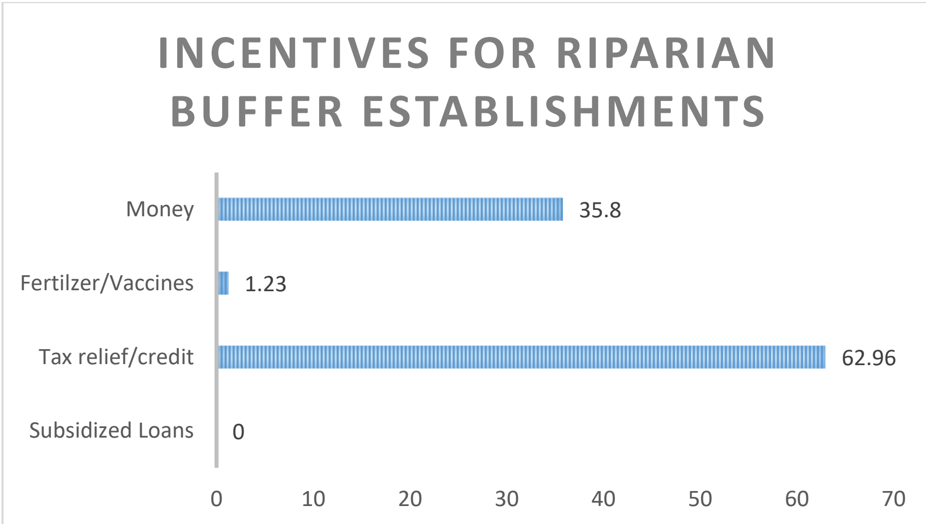

Figure 11. Farmers' and landowners' choice of incentives for riparian buffer establishment. None of the respondents required subsidized loans as incentive for riparian buffer establishment.

Table 1. Patch metrics of ecological patches in sparsely and highly populated counties in Virginia. The top group consists of counties with high population densities while the group below represents counties with low population density in Virginia. Column headings 1,2,3,4 and 5 represent General, Moderate, High, Very High and Outstanding Ecological Patches respectively. Patch density represents the number of patches per area.

County	**Smallest Eco. Zone Area (sqm)**	**Largest Eco. Zone Area (sqm)**	**Mean Eco. Zone Area (sqm)**	**No. of Zones**	**Patch Density**	**1**	**2**	**3**	**4**	**5**
Arlington	10640.01	10640.01	10640.01	1	8.96^{-5}					1
Fairfax	21824.55	11109427.46	1148488.22	161	9.27^{-4}		6	7	30	118
Henrico	4420.91	7940697.13	1156671.80	184	0.0018	1	2	29	45	107
P. Will.	101.52	19877063.91	1505955.58	252	0.0017	3	13	21	47	168
Chest.field	42.27	56102959.79	2146423.90	320	0.0018	4	8	20	66	222
Bland	33890.58	83251576.80	8341799.70	121	8.26^{-4}	8	9	14	21	69
Craig	647.48	88937080.31	11094030.77	96	7.05^{-4}	18	10	6	11	51
N.Hm	1577.24	102225325.17	2634284.16	148	4.54^{-4}	2	1	11	19	115
Bath	12.21	147980073.69	18289012.65	103	4.61^{-4}	16	17	14	14	42
Highland	34829.94	119701257.15	9273585.95	138	7.86^{-4}	7	10	16	27	78

Appendix F

VIRGINIA TECH

Are you a land owner?

- Yes
- No

Riparian buffers are vegetated or forested areas near waterbodies that improve the quality of water. On a scale of 1 to 5 (most valuable), how valuable are riparian buffers?

- 1
- 2
- 3
- 4
- 5

If you have or had a stream on your land, will you be willing to keep a 100 feet buffer around it?

- Yes
- No
- Maybe

Which of the following incentives for keeping a riparian buffer on your land would you prefer?

- Subsidized loans
- Tax relief/credit
- Fertilizer/Animal vaccines
- Monetary compensation