Land Use and Land Cover Change in the Crown of the Continent Ecosystem, Montana, USA from 1992-2011

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Abstract

In recent decades land use and land cover change (LULCC) has occurred throughout the Intermountain West. The Crown of the Continent Ecosystem (CCE) extends along the Rocky Mountains adjacent to the Canada-U.S. International border. In the U.S. portion of the CCE, located in northwestern Montana, development has increased since the 1990s, largely because of urban to rural migration. The CCE has become an amenity-based destination, which in turn is likely to threaten its terrestrial and aquatic ecological diversity (Quinn and Broberg 2007). Specifically, development pressures on private lands surrounding federally protected lands, are intensifying and thus threatening core habitat of native species and connectivity of forested areas. By characterizing the spatial and temporal patterns of LULCC, we can better understand landscape-scale changes influenced by human-environment interactions.

Using National Land Cover Database (NLCD) products, I identified areas that have experienced land cover change for three time periods: 1992-2001, 2001-2006, and 2006-2011. Additionally, I used case studies to further investigate LULCC in the study area. The findings suggest that the highest rates of development in proximity to Glacier National Park were dependent on existing urban land cover, meaning existing roadway infrastructure and established urban areas saw the greatest urban development. Additionally, communities adjacent to Glacier National Park were hotspots for urban development. Based on the results, areas in proximity to federally protected lands are likely to experience continued urban intensification over the next few decades.

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Chapter 1: Introduction

1.1 Introduction

U.S. nationally designated (federal) protected lands and their immediate surroundings have experienced high rates of regional land use change during the last several decades. Throughout the U.S., agricultural land is decreasing in rural areas, as popular recreational outlets and associated residential development expands (Riebsame, Theobald et al. 1996). Across the U.S., housing density, one measure of land use intensity, increased by nearly 750% on lands surrounding federally protected lands between 1940 and 2000 (Hansen, Piekielek et al. 2014). Rural housing development has become more common in some regions, and now presents itself as an obstacle to conserving native biodiversity because it converts formerly undeveloped areas into privately managed, developed land, which in turn impedes broad scale conservation measures. Low-density, rural land development has immense impacts on the landscape because it devotes a large amount of resources to service relatively few people. Further, this type of development occupies nearly 15 times the area than more densely urban areas (Brown, Johnson et al. 2005), reducing the total amount of road-less areas (Goetz, Jantz et al. 2009), and subjects the surrounding areas to further development and other human-induced land cover changes. Especially in rural areas, impervious cover tends to spread from existing zones impervious surface cover. Development pressures and land use and land cover change (LULCC), particularly in rural areas surrounding federally protected lands, are intensifying and thus threating core habitat and connectivity in the conterminous U.S. As development intensifies core habitat areas are likely to decrease in size, number and quality. As a result, these ecosystems will be increasingly more difficult to protect in their intact, natural state (Goetz, Jantz et al. 2009).

Mountainous areas within the western U.S. increasingly face the pressures described above. Within the last few decades, low density residential development has been occurring at a rapid pace in the Intermountain West, defined as the region made up of Arizona, Colorado, Idaho, Montana, Nevada, New Mexico, Utah, and Wyoming (Alm and Witt 1995). This form of development, typically characterized by isolated areas of residential, commercial centers, and corporate campuses (Zipperer, Wu et al. 2000), has resulted in extensive land use and land cover change across the Intermountain West region (Riebsame, Theobald et al. 1996, Rasker and Hansen 2000). More recent development over the last few decades differs from the previous movement of people into mountain regions, because it is driven by a high valuation of environmental and recreation amenities (Price, Moss et al. 1997, Glorioso and Moss 2007), rather than from resource extraction. Additionally, rural urban development is the fastest growing form of land use in the United States (Brown, Johnson et al. 2005).

The Crown of the Continent Ecosystem (CCE) extends along the Rocky Mountains from the U.S. into Canada. The CCE is one of two intact ecosystems remaining in the contiguous U.S. containing vast blocks of remaining intact wildlands in North America. Additionally, the CCE is home to a vast amount of terrestrial and aquatic ecological diversity as well to a relatively intact native biodiversity (Quinn and Broberg 2007). Multiple land agencies are represented within the CCE, including the National Park Service, the U.S. Forest Service, and the Bureau of Land Management, among other federal, state, and nongovernment agencies. Nearly 80% of the CCE is public land, much of which is protected by either federal or state agencies (Long 2007). Glacier National Park and the Bob Marshall Wilderness Complex are federally protected lands comprising the ecologically intact core of the U.S. portion of the CCE. National Forests in the

CCE alone receive over 3 million visitors annually, while Glacier National Park receives nearly 2 million visitors each year (Carolin, Gniadek et al. 2007, Long 2007).

Geographic scale relative to a specific region can offer a new way to conceptualize the implications of mountain development (Riebsame, Theobald et al. 1996). In the U.S. portion of the CCE, located in northwestern Montana, urban development has increased since the 1990s, largely because of urban to rural migration, in part due to the ecological and social amenities offered by mountainous areas. Studies have reported increases in urban development and land use change, but none have reported their full spatio-temporal extent at fine spatial scale within the CCE. The CCE has become a popular area for amenity-based recreation with its attractive mountain culture and recreation opportunities on ample public lands. As low-density residential development increases in areas adjacent to federally protected lands, development and accompanying land uses have the potential to adversely affect ecosystem function and integrity (Hansen and Defries 2007). Thus, as development pressures increase in the CCE and land use changes, the connectivity of core habitat can be subjected to accommodate increasing human demands. Recent development adjacent to federally protected lands has spread without regard for the potential impacts of land use and land cover conversion.

A more recent wave of residential and urban development in the Intermountain West has been made possible by ease of transportation and technology (Price, Moss et al. 1997), resulting in development of mountain-adjacent areas. The more recent migrants to the CCE, with different economic pursuits than those previously drawn to the CCE for the extractive industry, are partially responsible for the change to consumer-driven development patterns. A higher demand for outdoor, low impact recreational opportunities is accompanied by a younger, more homogenized, higher paid demographic shift. With this shift, low density residential development has increased in the last 30 years especially in the Intermountain West (McCool, Adams et al. 2006). The demands of the new migrants are typically quite consumptive and have been shown to have major and irreversible impacts on ecosystem integrity. Prato, Fagre et al. (2007) have confirmed ecological degradation of the CCE's natural landscapes as a result of growth and development on the private lands surrounding the protected and scenic lands of Glacier National Park.

1.2 Study Significance and Research Objectives

Land use change, and agriculture and urban development in particular, have a pronounced impact on ecosystem composition, habitat, and function (Prato 2012). In the CCE, development across the landscape as well as adjacent to federally protected lands has increased over the past few decades. Research to assess LULCC is necessary as urban development continues to increase and contribute to the overall built environment of the CCE. Long-term national-level assessment of land-use data has been fundamental to landscape-scale ecological studies. Yet, there exists a need for finer-grained information regarding land use patterns and land cover change (Brown, Johnson et al. 2005), which can help to provide insight into the effects caused by changing land uses. Thus, understanding the type and trajectory of LULCC over time and space is crucial to ecological conservation.

The goal of this research is to characterize observable changes in land use and urban area cover from 1992 to 2011 throughout the U.S. portion of the CCE. I use a LULCC approach to examine patterns of land cover change at a fine spatial scale, especially in rural areas where urban development is present. Specifically, I characterized observable LULCC from 1992-2011 using NLCD throughout the U.S. portion of the CCE with a focus on change trajectories of the entire CCE, and then on case study areas in close proximity to protected lands, including Glacier National Park and the USFS Bob Marshall Wilderness.

Specific objectives in support of this goal are to:

1. Perform accuracy assessments of the Landsat-based National Land Cover Database (NLCD) change from 2006-2011 for the entire CCE study area;
2. Characterize spatio-temporal patterns of LULCC within the CCE from 1992-2011; and
3. Use case studies to analyze landscape pattern change in areas where urban development and LULCC has occurred in close proximity to federally protected lands.

Research across the U.S. over the past two decades has reported land use change from agriculture to urban (Zipperer, Wu et al. 2000, Maestas, Knight et al. 2003, Prato 2012). Thus, I investigated agricultural land as a potential segue to urban succession, and quantified agricultural land cover susceptibility to urban development in various case studies. This study also evaluated whether agriculture abandonment for urban development is rapidly occurring in the CCE, and whether agriculture abandonment has led to other forms of LULCC.

Chapter 2: Theoretical Framework and Background

2.1 Introduction

This study approaches land use and land cover change in the CCE from a landscape ecology perspective. Although this study does not investigate amenity migration per se, amenity migration is a likely driver of landscape change in this region, based on reports from previous studies (Glorioso and Moss 2007, Prato and Fagre 2010, Silberman and Rees 2010). Thus, this section incorporates discussion on landscape ecology, the function and management of federally protected lands, and land use, for the purpose of examining landscape change and changing human-environment interactions in the region.

2.2 Landscape ecology

Landscape ecology is the science that examines the spatial pattern of multiple ecosystems in heterogeneous landscapes (Botequilha, Miller et al. 2006). Landscape ecology addresses the study of landscape patterns, the interactions among patches within a landscape mosaic, and how these patterns and interactions change over time (McGarigal and Marks 1995). A discussion on landscape ecology, however, is first dependent on a sound definition of the term 'landscape'. McGarigal et al. (1995) suggest that though numerous definitions of the term landscape exist, the definition should foremost be meaningful and relative to the context of the study. But landscape ecologists widely accept that landscapes are heterogeneous areas of land composed of interacting ecosystems that repeat in similar form throughout (Forman and Godron 1986), and are composed of patch mosaics that exist on a hierarchy across numerous spatial scales (Urban, O'Neill et al. 1987).

Landscape ecology offers both a conceptual and theoretical basis for understanding landscape structure, landscape function, and landscape change. Landscape function, or the interactions among spatial elements, is important when used in a landscape ecology framework. However, most relevant to this study is the concept of landscape change, which refers to the alteration in the structure and function of the ecological mosaic over a given time period (McGarigal and Marks 1995), and thus includes the development and dynamics of spatial heterogeneity (Risser 1984, Pickett and Cadenasso 1995). Importantly, landscapes are composed of more than just biophysical elements (Botequilha, Miller et al. 2006), such as human components. The interaction between nature and culture or behavior, for example, subsequently influences landscape function and is also a component of landscape composition.

Further, landscape ecology offers a useful applied framework for managing landscapes (Botequilha, Miller et al. 2006) in particular public lands and federally protected lands. Goetz (2009) analyzed core habitat areas in a landscape context using connectivity metrics to show that federally protected lands are often identified as core habitat, but are frequently isolated. The study stratified the core habitat areas based on land ownership and management. Isolated core habitat can prove less effective to species conservation and species resiliency from a conservation perspective. Riebsame and Theobald (1996) used regional, landscape, and site specific scales to evaluate sprawling land use conversion in rural, mountainous areas in the Colorado Mountains. They evaluated the landscape effects associated with the affluence of mountain development and the emergence of far-reaching rural sprawl. The framework used in this study exemplifies how planning focused at the landscape-scale can better accommodate changing land use trends.

An additional study performed by Brown et al. (2005) used an ecoregion classification as a framework to understand the magnitude, direction, and distribution of dominant spatial and temporal land use changes across the U.S. from 1950 to 2000. They were able to identify land use trends and interpret how development patterns correlate to different ecoregions. In the context of spatial distributions, this type of analysis can help to understand how landscape flows move differently throughout the landscape depending on what land cover types are adjacent or near one another. Further, researchers in the Greater Yellowstone Region conducted analyses to understand the relative influence of ecological, amenity, social, and economic variables on rural population growth in the states of Idaho, Montana, Wyoming, and the Greater Yellowstone Region (Rasker and Hansen 2000). Their findings add to the body of literature that attempts to explain why rural communities in the West have seen significant growth, with hope that land managers, researchers, and scientists can better understand the link between environmental conservation and development.

2.3 Federally protected lands and their ecosystem context

Federally protected lands are designated sea and land areas which can include strict nature reserves, wilderness areas, national parks, and management areas (DeFries, Hansen et al. 2007). Federally protected lands are any area managed by the federal government which prohibits incompatible land uses for the preservation of natural ecological processes (Possingham, Wilson et al. 2006). Traditionally, federally protected lands were created to conserve iconic landscapes and seascapes and to provide habitat for endangered wildlife (Watson, Dudley et al. 2014). Also, federally protected lands were established under a regime of dual mandate between species-landscape protection and public use (Watson, Dudley et al. 2014). The initial establishment of federally protected lands was founded on the principle that federally protected lands would protect natural features and wildlife in areas viewed as having little potential for economic use (Phillips 2004). However, federally protected lands may not be safeguarding core habitat effectively as land use and the demand for resources on rural landscapes intensifies. The landscape ecology framework for the study of changing landscape structure, function, and change applied to the management of federally protected lands can offer a magnitude of benefits to these ecosystems.

Federally protected lands are needed to abate the pressures of increasing human demands while balancing ecosystem function. For example, they can help to lessen the overall impacts on the greater ecosystem by spatially balancing concentrated land uses. The greater ecosystem provides the context for federally protected landscapes and is often defined by ranges of a particular species, hydrologic boundaries, or other ecological attributes, but may be defined by a number of ecologically relevant variables (DeFries, Hansen et al. 2007). However, it is common that the protected area boundary does not completely encompass the full gradients of biophysical factors within the greater ecosystem (DeFries, Hansen et al. 2007). Consequently, the greater ecosystem often lies outside of the boundaries of the protected area and is therefore subject to private land use intensification which have implication for landscape structure and function.

The relative size and placement of a protected area within its greater ecosystem is potentially more important than the absolute size of a protected area in regards to ecosystem function (Watson, Dudley et al. 2014). Watson et al. (2014) also state that the relative importance of ecological mechanisms varies with the ecological characteristics and socioeconomic setting. DeFries et al. (2007) state that delineating the placement of the protected area within the biophysical gradients of the greater ecosystem is key for identifying movements of species, critical habitats and other ecological interactions.

Land use intensification adjacent to federally protected lands has only further contributed to the pressure on ecosystems and landscapes globally. Tourism inside federally protected lands gained popularity during the middle of the twentieth century (Zeiger, Caneday et al. 1992), and an increase in housing density surrounding protected area-centered ecosystems occurred between 1940 and 2000, especially within the contiguous United States (Hansen, Piekielek et al. 2014). For example, DeFries et al. (2007) noted that federally protected lands in affluent countries will continue to be magnets for second homes and tourism. Watson et al. (2014) argue that protected area effectiveness in recent years is being largely affected by "defunding, downscaling, and delisting of federally protected lands" (Watson, Dudley et al. 2014, p. 67). They advocate that protected area performance can be improved by better management practice, stating that when federally protected lands are managed to target specific occurrences that federally protected lands can abate and deliver essential ecosystem services.

The extent and magnitude of resource use in and around a protected area will determine which management approaches might be most effective to balance human demands with ecological function. DeFries et al. (2007) state that in more affluent areas, management that deflects development away from critical habitats might be achieved through existing institutions for zoning and economic incentives. They further state that management should aim to identify "small loss-big gain" opportunities, in which ecological functioning of the protected area might be maintained with minimum negative consequences for human land use as well as ecosystem well-being. Identification of "small loss-big gain" opportunities is dependent on a detailed understanding of the ecological and socioeconomic settings of the surrounding area. Ultimately, human needs and desires for land and other resources determine both the pressures on federally protected lands and the management of them (DeFries, Hansen et al. 2007). Federally protected lands have the potential to achieve biodiversity conservation, continue to deliver essential ecosystem services, and protect relatively intact landscapes for future generations if managed in a relevant spatial context.

2.4 Amenity migration in the CCE

Amenity migration refers to the relocation of individuals based on their desire to live near areas of perceived natural beauty (such as mountains and beaches), or areas rich in environmental and cultural resources (such as areas with interesting architecture, cultural facilities, and the presence of connected neighborhoods). Amenity-rich destinations tend to offer ample recreational outlets, are easily accessible by either car or airplane, and provide access to commodities and resources through modern technology (Price, Moss et al. 1997, Glorioso and Moss 2007, Prato and Fagre 2010). As previously noted, federally protected lands are considered an attractive amenity that draws amenity migrants to rural regions. Thus, public land adjacency is an attractive real estate trait because of the ease of access to recreational activities and opportunities (Riebsame, Theobald et al. 1996, p. 403).

Amenity-rich areas are regions that attract residents because of their “comparatively high concentrations of Earth’s remaining environmental quality and cultural distinctiveness” (Glorioso and Moss 2007, p. 2). Yet, locations with perceived high-amenities are not confined to topographically significant ecosystems. In fact, Price et al. (1997) state that amenity migrants are attracted not by the physical resources which can be extracted, such as minerals, trees, or grass for livestock, but by a vision of a quiet life, clean air, and a more pleasant environment. However, mountainous areas within the western U.S., until recently, have long been sheltered from human occupation due to their inaccessibility throughout most of human history (Prato and Fagre 2010). Rasker and Hansen (2000) found that more mountainous counties experienced higher population growth rates than neighboring counties with widespread plains in the Greater Yellowstone Ecosystem (GYE). Additionally, population growth in the rural counties of Montana, Idaho, and Wyoming significantly correlates with forest cover, high variation in topography, maximum precipitation, and the degree to which the land is in some form of protected status (Rasker and Hansen 2000).

Prior to the 19th century, the economy of the CCE was based largely on extraction (Prato and Fagre 2010). The founding communities in the Rocky Mountains were prompted by the discovery of gold, silver, copper, and other metals. The practice of ranching in high mountain basins to supply food and animal power was in support of the timber extraction industry (Silberman and Rees 2010). Since then, the Intermountain Western States, specifically those that are within or adjacent to the Rocky Mountains, have experienced the greatest population increases since 1970 (Riebsame, Theobald et al. 1996). The northern forested areas of the Rocky Mountains experienced rapid population increases in rural areas between 1970 and 2000, while most of the U.S. was experiencing suburban growth (Brown, Johnson et al. 2005), illustrating the "increasing importance of amenity-driven development that has been evident since 1970" (Brown, Johnson et al. 2005, p. 1858). Beginning in the late 1980s, single family homes were becoming more common in rural areas – an indicator of both amenity migration and residential development (Riebsame, Theobald et al. 1996).

The development of rural landscapes by amenity migrants has the potential to shift the focus of the local economic base from one dominated by extraction to one that emphasizes recreation, further promoting a continued in-migration (Glorioso and Moss 2007). The more recent migrants, those driven by natural and cultural amenities, come from different economic backgrounds and have different priorities and economic demands. Residential and urban development in rural areas has substantial socio-cultural, and economic effects on mountainous regions and amenity-rich areas alike. Many amenity migrants tend to have greater discretionary wealth and time than those individuals employed by extractive industries (Glorioso and Moss 2007). An estranged economic agenda, combined with the fact that new migrants tend to be more educated and occupy jobs in the service sector, versus extractive or agricultural industries (Rasker and Hansen 2000), puts an undeniable pressure on amenity regions. Town populations in counties with recreational opportunities, and thereby an above average rate of migrants, differ in their desires to sustain small-town community life and pastoral landscapes, or support the economic growth that owes itself to the recreation economy (Muller, Yin et al. 2008).

Increased affluence, technology, and population density all contribute to greater rural development in formerly wild areas (Hansen and Defries 2007). Glorioso and colleagues (2007) found that while natural and cultural amenities help to motivate amenity migrants, the late modern political-economy helps to facilitate this migration to formerly remote mountain regions. As previously mentioned, an expanded range of attractions and dependable accessibility to commodities and resources, despite living in relatively remote areas, tends to further promote development of the rural landscape (Price, Moss et al. 1997). Meanwhile, technology and reliable transportation provides a comfortable living environment, which enables amenity migrants to maintain their current lifestyles while living in more remote, rural areas (Price, Moss et al. 1997). This not only has implications for local communities and long term residents, but can have significant consequences for land use and land cover conversion in the region.

Price et al. (1997) state that amenity migrants move to places where they were once previously a temporary visitor or tourist. This trend may explain relationships between accessible travel and future susceptibility of an amenity-rich mountainous area, or certain population traits as a determinant of future economic growth (Rasker 2006). In a report released by the National Park Service, Glacier NP received nearly 2.4 million visitors in 2015 which was the busiest year on record (Annual Park Recreation Visitation – Glacier NP). Meanwhile, the numbers of visitors and amenity migrants to the CCE has increased over the past few decades (Prato and Fagre 2010).

2.5 Land use

Urban development set in a rural setting places great strain on the resources needed to service a less concentrated population (Price, Moss et al. 1997). Amenity migrants' expectations often include access to resources that parallel those of urban environments. Additionally, urban land use is often a low percentage of total catchment area yet exerts a disproportionately large influence across the landscape (Paul and Meyer 2008). Ease of transportation and development adjacent to both pre-existing development and amenity rich areas is common in low density residential areas (Leitão, Miller et al. 2012), which helps to explain why residential building sites tend to press up against public land boundaries, and thus against the most wild and federally protected lands.

Urban areas and development subject rural areas to further development, fragment the ecosystem, and obstruct migratory patterns of local species through the construction of road networks and the fencing of property or highway infrastructure. This is of great concern for mountain ecosystems, especially those of the CCE, which are home to endangered species, threatened species, and apex predators. The conflicting issue regrading land use arises when building and development begin to encroach on habitat used by native populations of mammals, birds, and fauna. Roads create access for humans to engage in extraction, recreational, or residential development activities (Leitão, Miller et al. 2012). The effectiveness of reserves, as well as various aspects of biodiversity, are affected by fragmentation and habitat loss resulting from residential development (Hansen, Knight et al. 2005). Additionally, development further promotes the reduction of road-less areas, increases runoff, and contributes to the positive feedback between developed land and future settlement. These consequences of urbanization and the building of roadway networks are worsened by the inefficiency of rural development in supplying resources to meet the demands of the human population.

Urban development is one form of land use that has a pronounced effect on the greater ecosystem. Additionally, impervious cover tends to increase in areas where impervious cover is already established (Goetz, Jantz et al. 2009). Thus, acting like a positive feedback system, development continually promotes ecosystem fragmentation. Not only do roadways obstruct migratory and dispersal patterns of plant and wildlife, but wildlife boundaries pose concern for both plant and animal species. Political borders are porous to plant and animal wildlife yet pose a threat to wildlife management. In an area where the greater ecosystem overlaps with privately managed lands, human impact on the ecosystem and the species within it can be detrimental. The crossing from one designated area to another by plant and animal species has multiple interpretations depending on the context of the managing group concerned (conservation, land management, and political). Additionally, vehicles act as vectors of long-distance seed dispersal with implications for non-native plant invasions (Taylor, Brummer et al. 2012). Further, impacts of transportation corridors on surrounding habitat often reaches far beyond the edge of the corridor with the greatest range and frequency of seed dispersal of non-native species occurring on grasslands versus forested areas (Hansen and Clevenger 2005). Roads can create artificial edges that opportunistic wildlife species exploit, and can be a major cause of morality when individuals attempt to cross roads (Leitão, Miller et al. 2012). Furthermore, expansion of less concentrated, rural areas has been widely recognized as the primary factor influencing the management of U.S. National Forests (Theobald and Romme 2007).

Urban landscapes are a complex mosaic of human modification and built structure. The likeliness of forest clearing increases as overall developed land increases (Brown, Pijanowski et al. 2000), meaning that there exists a strong relationship between already developed lands and the probability that adjacent, undeveloped land will soon become developed. However, even as development negatively correlates to forest cover, this is not a direct causation (Brown, Pijanowski et al. 2000).The vulnerability of an area to development may be due to the fact that an area with development is easily accessible. "The socioeconomic fabric of surrounding human communities probably also influences the vulnerability of federally protected lands" (Hansen and Defries 2007, p. 985). An area's vulnerability to land use is dependent on a number of factors. A few of these factors include the ecological properties of the surrounding ecosystem, the type and intensity of land use conversion, as well as the extent in which the human-environment relationship continues (Hansen and Defries 2007). It is important to note that the human-environment relationship can be defined at a very small scale or even globally, seeing as global climate change has its effects on many ecosystems as does the attitudes and actions of the permanent and temporary human populations. Additionally, regions where forest patches are small tends to relate to regions where little forest actually exists (Fahrig 2003). Again, the vulnerability of an area to development is dependent on how easily accessible that area is, and the socioeconomic fabric of the human communities surrounding it.

Agriculture and other anthropogenic activity extends to the stream margin when natural riparian forest is removed. Under this condition, streams are usually warmer during summer and receive fewer energy inputs as leaf litter and primary production increases (Quinn 2000). Additionally, bank stability decreases. Agricultural land use degrades steams by increasing nonpoint inputs of pollutants, impacting riparian and stream channel habitat, altering flows (Allan 2004)(p. 264). Deciduous forest land cover has far less runoff than developed high intensity land cover, but this also depends on the combination of soils, chemical composition of agricultural and lawn care, and land cover type. Row crop and other forms of intensive cultivation strongly affect stream condition, while influence of pasture agriculture may be less pronounced (Meador and Goldstein 2003). Land use influences runoff which has serious ecological implications.

The total extent of the wildland-urban interface, or the area between unoccupied land and human development, is likely to continue to increase through 2030 with the greatest expansion predicted to occur in the Intermountain Western States (Theobald and Romme 2007). Additionally, management or lack thereof, will have an impact on the future state of this ecosystem among many other ecosystems around the world with similar threats and composition. As decentralization of employment increases, the amount of rural land accessible to these new workplaces and available for development increases exponentially, thus allowing further suburbanization (Zipperer, Wu et al. 2000). Alan (2004) also states that legacy effects are the consequence of disturbances that continue to influence environmental conditions long after the initial appearance of the disturbance. Gradients of anthropogenic land use are frequently superimposed on an underlying gradient in parent geological material, soil type, topography, and other features of the natural terrain. Today however, the CCE is being exploited faster than ever before with technological advances. Terrain, topography, and other once prohibiting natural features are now desirable characteristics of second home dwellers. Allan (2004) argues that for management and restoration actions to be effective, we must diagnose cause as well as assess harm. Zipperer et al. (2000) warn that the edge city may have greater effects on ecological and social resources than previous patterns of suburbanization. Yet, even suburbanization is an enormous threat in itself. Unlike the older central city, which may experience a revival because of interesting architecture, connected neighborhoods, and the presence of government centers, universities, and cultural facilities, inner suburban revival is unlikely (Zipperer, Wu et al. 2000). Additionally, the long-term sustainability of suburban development, especially second home amenity apartments that lack desirable context and connectivity, is questionable.

Chapter 3: Study Area and Methods

3.1 Study Area

The study is the U.S. portion of the CCE, a North American Rocky Mountain ecosystem extending from the Elk and Highwood Rivers in British Columbia and Alberta south into northwestern Montana's Blackfoot River Valley (Figure 1). It is a distinctive, emerging locale of amenity migration characterized by a complex mosaic of natural, public and private land (Prato and Fagre, 2007). The U.S. portion of the CCE (~47,200 km^2) is comprised of 80% federal and state public land, including Glacier National Park, lands managed by the USDA Forest Service (including Wilderness Areas and National Forests), the Bureau of Indian Affairs, and Montana State Forests. I focus on the U.S. portion only, given data availability and data format inconsistencies across the border. Private land is largely composed of agriculture, or owned by timber companies. In Flathead County alone, employment growth is found primarily in trade and service sectors, as well as in real estate and construction. Longstanding industries, including wood products manufacturing and agriculture, are now generally on the decline (Prato and Fagre, 2007).

In addition to jurisdictional complexity, tremendous terrestrial and aquatic ecological diversity also characterizes the CCE, as indicated by the number of Environmental Protection Agency-designated Level IV Ecoregions found in this region (Omernik, 1987). Topographic complexity and relief (~1000 m on the Blackfeet Indian Reservation to 3194 m on Mount Cleveland in Glacier National Park), contribute to the region's extensive and intact native biodiversity (Quinn and Broburg, 2007) given their relationship to meso- and micro-climatic variation (Prato and Fagre 2010). Prairies and foothill vegetation along the eastern Rocky Mountain front transition into a patchwork of montane mixed deciduous and coniferous forests at higher elevations, eventually to be replaced by subalpine and alpine plant species at the higher elevations. This ecological diversity translates to habitat for 65 mammals native to the CCE, a grizzly bear and wolf recovery area, North America's most diverse ungulate species composition, and elusive species such as the wolverine, which require large areas of undisturbed core habitat (Long 2007).

The CCE offers a vast array of recreational opportunities; its National Forest areas alone receive over 3 million visitors per year (U.S. Forest Service, 2008). The region also includes four-season recreation resort communities in the Flathead Valley, such as Whitefish, MT — a GNP gateway community offering world-class amenities. Overall, from 1990-2010, the CCE population grew 35.5%, compared to 23.6% for Montana as a whole (U.S. Census Bureau 2010). An estimated 75% of this growth has been absorbed by low density residential development, resulting in rapid land cover conversion in rural areas (McCool and Adams, 2007). Flathead County is the second fastest growing county in Montana. A forty-five percent rise in total personal income (1987-1997) has paralleled this growth. Given the jurisdictional, ecological and topographic complexity of the CCE, scientific knowledge is incomplete and data acquisition about the region is challenging (Prato and Fagre, 2007).

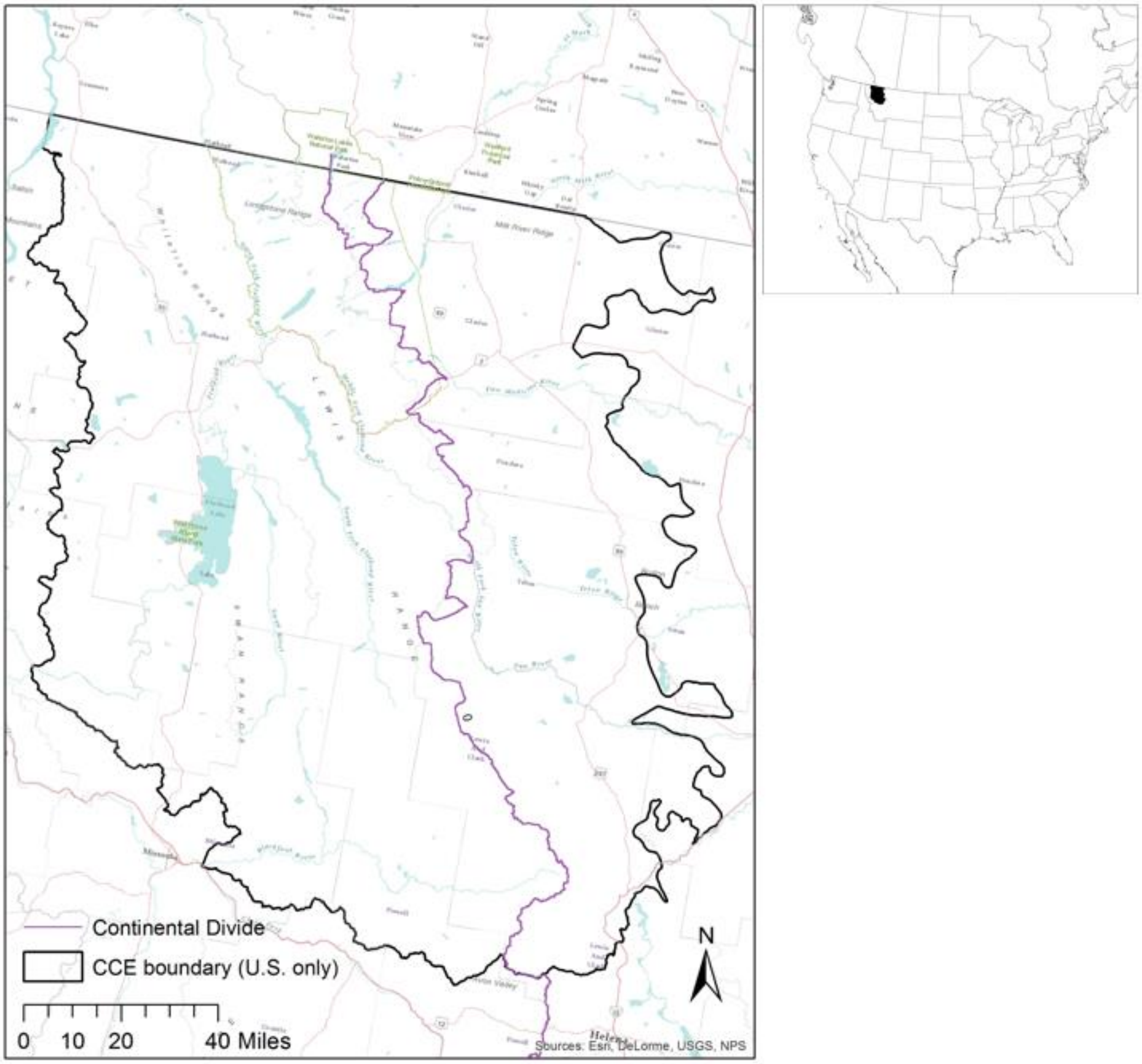

Figure 1. Location of the CCE, which is located along the Northern Rocky Mountains adjacent to the Canada-U.S. international border.

3.2 Methods

3.2.1 Data

The National Land Cover Dataset (NLCD) products have been increasingly used in LULCC studies (Vogelmann, Howard et al. 2001, Homer, Dewitz et al. 2007, Fry, Xian et al. 2011). In this study, I use NLCD products produced by the Multi-Resolution Land Characteristics Consortium (MRLC) for the conterminous United States from 1992-2011, which has a 30-meter spatial resolution. Using the NLCD 1992/2001 Retrofit Land Cover Change Product I created a 1992 land cover map of the study area. For 2001-2011, I used NLCD 2001, NLCD 2006, and NLCD 2011, which includes 16 land cover classes, which I then aggregated into four classes relevant to the Google Earth accuracy assessment, the temporal (Objective 2), and spatial (Objective 3) aspects of this study (Table 1). I used these four land cover maps to perform an urban accuracy assessment for NLCD 2011, a land cover change accuracy assessment for NLCD 2006-2011, and to compare the spatio-temporal patterns of LULCC across 1992-2011 time period.

Per the MRLC website, the NLCD 1992/2001 Retrofit Land Cover Change Product provides land cover change information at the Anderson Level I classification scale (Anderson, Hargy et al. 1976), relying on decision tree classification of Landsat satellite imagery from circa 1992 and 2001. Since the NLCD 1992/2001 Retrofit Land Cover Change dataset had pre-grouped agriculture, forest, and urban land cover classes using a different classification scheme than that of the 2001, 2006, 2011 datasets, this dataset required some pre-processing to make it comparable to the others. Specifically, the 1992 dataset did not differentiate the open space land cover category present in the 2001, 2006 and 2011 datasets. The open space pixels from 2001 map were used to approximate the open space pixels in 1992. Such approach is questionable in identifying “true” open space class in 1992. An alternate method is to use raw Landsat images to re-build 1992 land cover map products, which require substantial time and effort for a stand-alone remote sensing research task. For all four land cover maps, we applied a Majority Filter (3 by 3 window) to remove the salt and pepper noise in the original land cover products.

Table 1. Reclassification scheme used in the land cover change analysis from 1992 to 2011.

Class (Level II)	**Name**	**Reassigned Name**
21	Developed, open space	Open Space
22 23 24	Developed, low intensity Developed, medium intensity Developed, high intensity	Urban
41 42 43	Deciduous forest Evergreen forest Mixed forest	Forest
81 82	Pasture/hay Cultivated crops	Agriculture

3.2.2 Land Cover Accuracy Assessment

OVERVIEW

Few studies have specifically assessed the accuracy of land cover change using NLCD, especially in mountainous areas where terrain shadowing, high relief, cloud cover, and limited field data can affect land cover classification and change analysis accuracies (Burnicki 2011, Wickham, Stehman et al. 2013). One known study by Wickham et al. (2013) assessed the accuracy of land cover change in the conterminous United States between 2001 and 2006 using NLCD 2001 and NLCD 2006. Thus, I used this opportunity to evaluate the usage of national scale products for local application in a mountainous region. The accuracy assessment was composed of a field component which assessed the accuracy of the NCLCD 2011 developed land cover classes, and an image comparison component which assessed the accuracy of land cover change using NLCD 2006 and NLCD 2011.

FIELD VALIDATION OF URBAN LAND COVER CLASSES

Field validation was performed throughout the months of June and July of 2015 for the four developed land cover classes using NLCD 2011. Prior to entering the field, I used NLCD from 2006 and 2011 to derive a raster land cover change map, which ensured that I was looking at only new urban development on the 2011 land cover dataset. Once areas of change were detected, each change location was assigned a central point using ArcMap 10.3 (ESRI, Redlands, CA) which I then converted to geographic coordinates and uploaded into the GPS unit for field navigation.

Once in the field, I randomly selected coordinates to serve as validation sites based solely on accessibility. Restrictions were typically a result of private land ownership limiting my access to some coordinates. A total of 89 coordinates were visited in the field, with a focus on field validating the following NLCD 2011 land cover classes: 21 (developed, open space), 22 (developed, low intensity), 23 (developed, medium intensity), and 24 (developed, high intensity) (Figure 2). Urban development to low, medium, or high intensity was typically from a lesser form of development (i.e. open space to developed, low intensity; or developed, low intensity to developed, medium intensity). This accuracy assessment served to justify the grouping of NLCD class 21 (developed, open space) as a separate entity from the other NLCD urban land cover classes, and also enabled a meaningful open space land cover category for subsequent analysis.

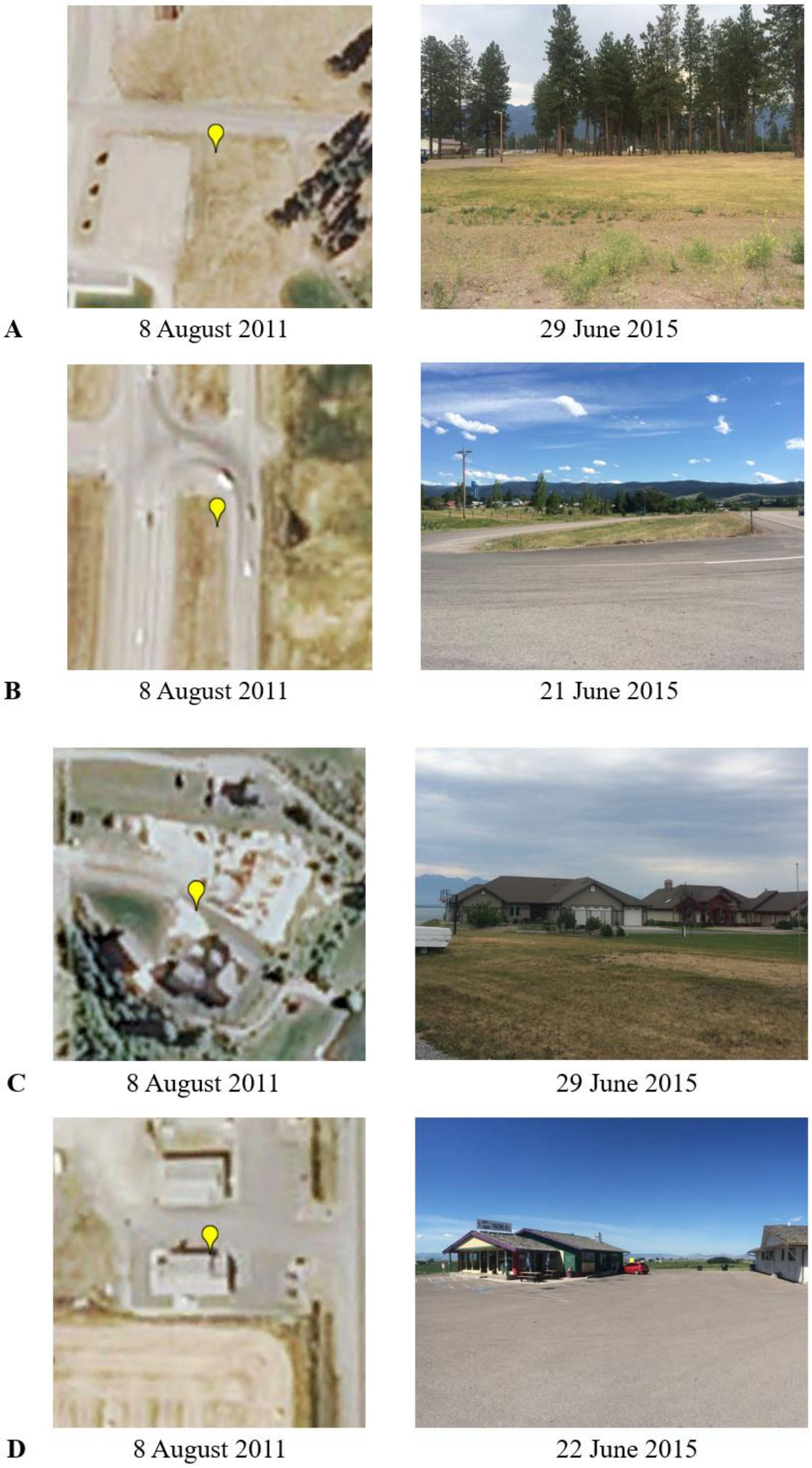

Figure 2. NLCD 2011 urban land cover accuracy assessment. First column = Landsat Imagery, derived from Google Earth from August 2011; Second column = on-site photo taken in June 2015. (A) NLCD Class 21 (developed, open space). (B) NLCD Class 22 (developed, low intensity). (C) NLCD Class 23 (developed, medium intensity). (D) NLCD Class 24 (developed, high intensity).

LANDSAT-BASED LAND COVER CHANGE ACCURACY ASSESSMENT

For the second accuracy assessment component, I evaluated the accuracy of land cover change from 2006-2011 using Landsat-derived imagery from Google Earth in addition to the land cover change raster I created. The land cover classes selected for the assessment of land cover change were forest to grassland, grassland to forest, agriculture to grassland, grassland to agriculture, and agriculture to urban. I selected these land cover change classes because I anticipated that these land cover types may be associated with anthropogenic land use change, such as deforestation, urban development, water diversion and dams, and agricultural intensity or abandonment (Lawler 2009). Additionally, I selected these land cover change classes because they reflect the dynamic nature of the CCE landscape. Landscapes, both natural and built, can have a number of land cover change trajectories, human and environment induced, thus observing change, especially change from grassland to forest and agriculture to grassland, can aid in the overall assessment of land cover change.

For analysis, each land cover class for NLCD 2006 and NLCD 2011 was aggregated to the Anderson Level I classification scheme prior to creating the land cover change map between the two sets of data. I omitted NLCD class 21 (developed, open space) from the urban classification for this portion of the accuracy assessment because of the low accuracy of the land cover class (Wickham, Stehman et al. 2013). Thus, the urban land cover class included NLCD classes 22 (developed, low intensity), 23 (developed, medium intensity), and 24 (developed, high intensity). Finally, for each of the change classes selected, the total area of land cover change occupied at least 0.01% of the study area (Table 2). Land covering 0.01% of the image guaranteed enough land area to perform a stratified random sample, and also ensured that none of the coordinates of the stratified random sample were directly adjacent to another random point during analysis.

Table 2. Land use and land cover change classes observed. The total land area (km^2) is at least .01% of the study area.

2006-2011 Land Cover Change Class	**Total Area of Change (km^2)**
Forest to Grassland	151,397
Grassland to Forest	12,597
Agriculture to Grassland	8,051
Grassland to Agriculture	4,046
Agriculture to Urban	1,019

I randomly selected a 30 point sample for each land cover change class to use in accuracy assessment. Pixels were selected by extracting the areas of one type of land cover change (i.e. grassland to urban) and assigning 30 random points over the entire area, to ensure that larger patches were not discounted or weighted less than smaller patches. Each point was then compared against high resolution digital imagery from Google Earth at two dates (Figure 3), which served as reference imagery and was used for the generation of a truth map (Bossler, McMaster et al. 2010). The pixel representative of the land cover conversion class of interest was then used for the error matrices analysis. Error matrices were generated manually to evaluate overall accuracy, omission error, and commission error (Congalton and Green 2008, Bossler, McMaster et al. 2010). Overall accuracy refers to the total number of correctly identified land cover change sites, divided by the total number of land cover sites evaluated; omission errors refers to the error of exclusion, or the probability that a certain land cover class in the map is in fact this class; and commission error refers to error of inclusion, or the certainty of which the user can expect the pixels classified to be true.

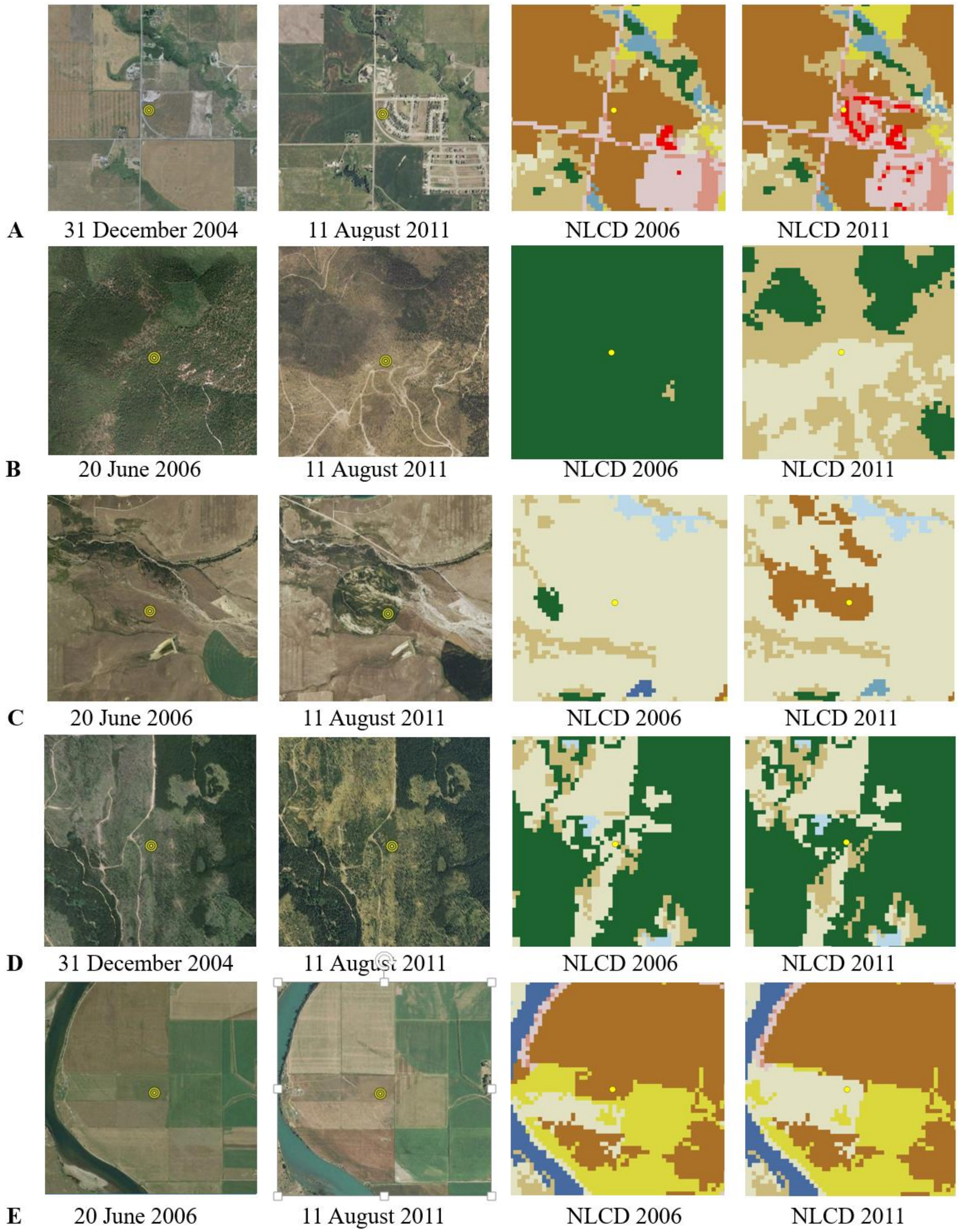

Figure 3. Google Earth accuracy assessment method. Landsat Imagery, derived from Google Earth from 2005 and 2011 (First and second columns, left to right respectively); Third column = NLCD 2006 map. Fourth column =NLCD 2011 map. (A) Agriculture to urban land cover change. (B) Forest to grassland land cover change. (C) Grassland to agriculture land cover change. (D) Grassland to forest land cover change. (E) Agriculture to grassland land cover change.

LAND USE AND LAND COVER CHANGE ANALYSIS

A change detection analysis between image pairs of different years quantified the type and trajectory of LULCC. Analysis of sequential NLCD aided in the characterization of the spatial and temporal patterns of LULCC from 1992-2011 (Objective 2). Two sets of images of the same scene but different dates (e.g. 1992-2001, 2001-2006, and 2006-2011) were used to assess change and to characterize the land cover conversion and land use change.

To perform the change detection, original NLCD classes were again regrouped into broader classes using the Anderson Level II classification scheme as a reference (Table 3). Land cover types meaningful to this portion of the research were agriculture, urban, forest, and grassland because I anticipated that these land cover classes may be associated with anthropogenic land use change, such as deforestation, increased urban land cover, and agricultural intensity or abandonment. Land cover classes were selected based on research documenting a shift from an extraction based economy to an amenity based economy in parts of the Rocky Mountain West (Riebsame, Theobald et al. 1996, McCool, Adams et al. 2006, Muller, Yin et al. 2008).

Table 3. Major land cover classes used for the LULCC assessment in the CCE from 1992-2011.

Class (Level II)	**Name**	**Reassigned Name**
21	Developed, open space	Open Space
22 23 24	Developed, low intensity Developed, medium intensity Developed, high intensity	Urban
41 42 43	Deciduous forest Evergreen forest Mixed forest	Forest
81 82	Pasture/hay Cultivated crops	Agriculture

For this study, forested areas were defined by combining the NLCD classifications for deciduous forest, mixed forest, and evergreen forest. Agriculture was defined by combining the NLCD classifications for pasture/hay and cultivated crops. Urban areas are composed of the three urban classes, low, medium, and highly developed intensity; and open space was kept as a singular classification unit defined only by NLCD 21 (developed, open space). Forested areas are of particular interest because they serve as a habitat for many local species, are a well-studied component of the urban-forest gradient, are an indicator of anthropogenic influence when in a state of deforestation or recent logging, and are one of the defining components of a high amenity area (Price, Moss et al. 1997). Observing both agricultural land cover, together with urban land cover can help to explain if agricultural lands are becoming abandoned for urban development. Finally, the fourth land cover class of interest that I selected was developed, open space. Observing open space and urban land cover helps to further access the urban development phenomena occurring on land adjacent to federally protected lands (Goetz, Jantz et al. 2009).

LANDSCAPE METRIC ANALYSIS

The third and final objective was to characterize LULCC in case studies in the CCE. The areas of Kalispell, Whitefish, West Glacier, and Polebridge were selected to use as case study locations for this study. I used Fragstats (McGarigal and Marks 1995) to quantify landscape structure and characterize the spatial composition of the four aggregated land cover classes (open space, forest, urban, and agriculture) at 1992, 2001, 2006, and 2011. I corroborated these landscape metric analyses with U.S. Census data to improve the understanding of all observed change.

Case study locations for landscape pattern analysis were selected based upon prior knowledge of the region, information provided in the literature, and the use of 2010 TIGER/Line shapefiles combined with block-group level population data from the 2010 U.S. Census to identify the most densely populated areas in 2010. Block-group level population data provides spatial detail regarding population statistics and enabled us to calculate the population density for a given area within the study area. Further, densely populated areas were selected because of the various outcomes associated with land use change due to the economic and demographic growth of the Rocky Mountain region (Vias and Carruthers 2005). Often, a greater population density at 2010 correlated with a smaller block group size; for that reason, I selected block groups completely contained with Flathead County, Montana, and Lake County, Montana.

The West Glacier and Polebridge study sites are located immediately adjacent to the federally protected lands of Glacier National Park, while Whitefish and Kalispell are located in its vicinity, but not adjacent (Figure 4). West Glacier is a known gateway community to Glacier National Park, while Polebridgge is an emerging amenity community. Whitefish and Kalispell were selected based on their relatively dense populations in 2010. Additionally, because Whitefish and Kalispell are relatively concentrated urban areas, I assessed urban sprawl over time by analyzing block groups on the periphery (termed “greater areas”) for each of these regions. Population characteristics at the block group level were taken into account during the selection of these two study sties; thus, analysis of landscape metrics for Whitefish and Kalispell was performed twice—once for the more concentrated, urban area, and again for the surrounding landscape which did not include the urban landscape (Table 4).

Development of the rural landscape adjacent to protected lands has accounted for much of the LULCC in the region, especially over the last few decades (Rasker and Hansen 2000, McCool, Adams et al. 2006, Rasker 2006, Glorioso and Moss 2007). Thus, I was interested to see if this trend was present in the CCE, and I was interested in the probable trajectory of land cover change. For the selection of case study sites I took into account how developed the area was according to NLCD 2011, and the degree to which the literature suggests the area was developed.

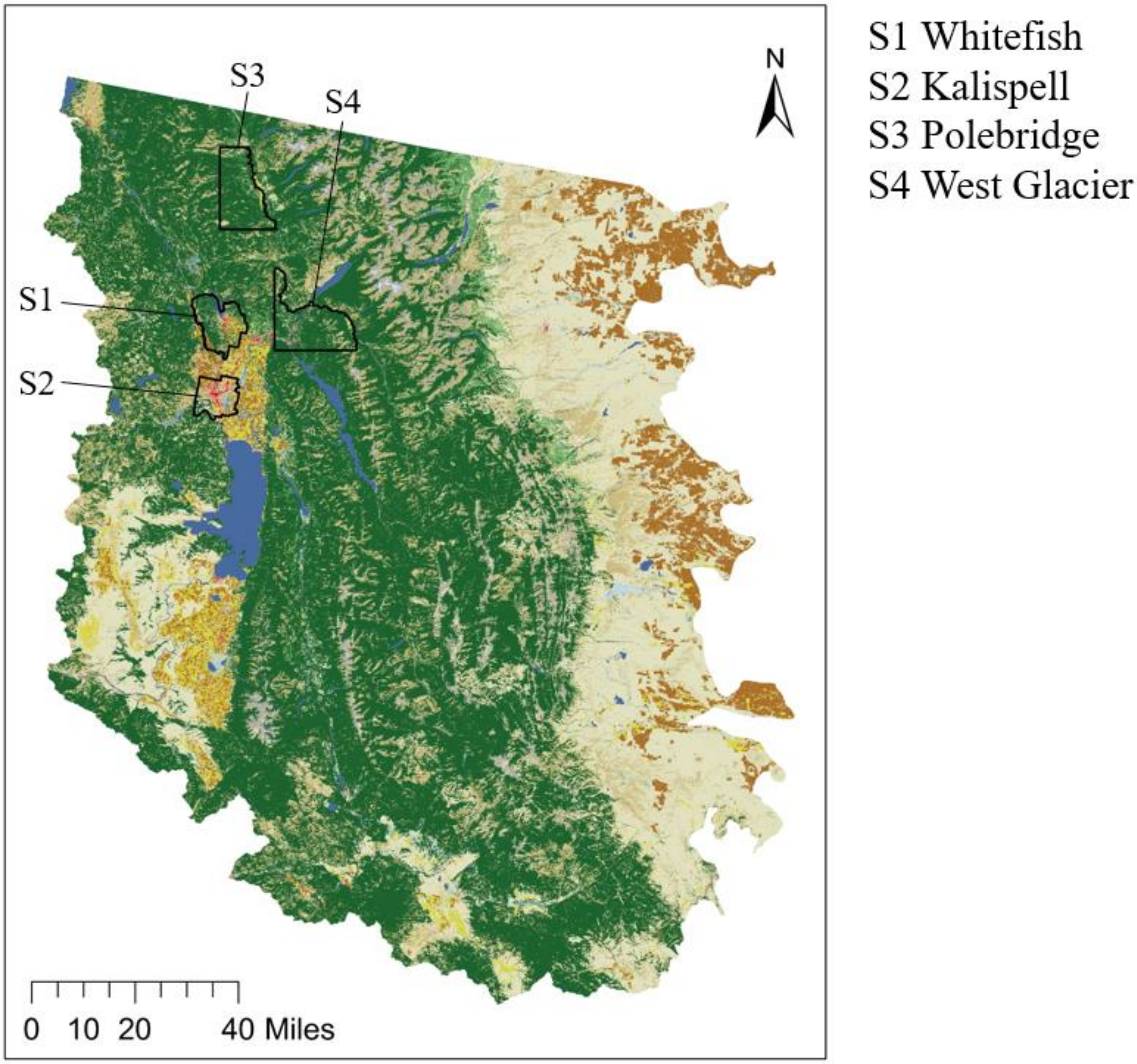

Figure 4. Study sites used for Fragstats analysis.

Table 4. Population characteristics of case study sites

Site	County	Population Density 2010	Population Density 2010 Standard Deviation	Total Land Area (km^2)	Percent of Flathead County	Smallest Population Density in a Block Group during 2010	Largest Population Density in a Block Group during 2010	Nearby amenities and attractions
Kalispell	Flathead	1.36	0.60	10.52	0.08%	0.631	2.69	Blacktail Mountain Ski Area
Surrounding Kalispell Area	Flathead	0.86	0.67	139.20	1.06%	0.049	2.69	Blacktail Mountain Ski Area
Whitefish	Flathead	0.68	0.34	9.06	0.07%	0.22	1.279	Blacktail Mountain Ski Area; Whitefish Mountain Ski Resort and
Surrounding Whitefish Area	Flathead	0.37	0.38	198.04	1.51%	0.019	1.279	Blacktail Mountain Ski Area; Whitefish Mountain Ski Resort

I used Fragstats v4.2.1, (McGarigal and Marks 1995), a landscape metrics quantification program to assess change in landscape structure of the CCE as a whole, and each of the study areas for each of the land use classes for the years 1992, 2001, 2006, and 2011. I analyzed raster imagery at the class- and landscape-levels. Class-level metrics were calculated for each land cover class (urban, open space, forest, and agriculture) in each of the defined study sties. Landscape-level metrics are calculated across the entire CCE study area with no delineation between individual land cover classes. Patch-level metrics are calculated on every patch of a specific land cover class; however I did not use this metric in the analysis. The metrics I used are: Class Area (CA), Largest Patch Index (LPI), Number of Patches (NP), and Mean Patch Area (Area_MN) (Table 5).

Table 5. Selected metrics for land cover classes, calculation and justification.

Metric	Calculation	Justification	Units
Class Area (CA)	Sum of the areas (m^2) of all patches of the corresponding type, divided by 10,000 (to convert to hectares). As CA approaches 0, the patch type becomes increasing rare in the landscape.	Helps ascertain how much of the landscape is comprised of a particular patch type.	Hectares
Largest Patch Index (LPI)	Largest Patch Index (LPI) = area (m^2) of the largest patch of the corresponding patch type / total landscape area (m^2), multiplied by 100 LPI will equal 100 when the largest patch comprises 100% of the landscape.	Provides a simple measure of dominance. At the class level, LPI quantifies the percentage of the total landscape area comprised by the largest patch.	Percent
Number of Patches (NP)	The total number of patches in the landscape.	The importance of a given patch for habitat conservation is dependent on the relationships with other patches of a similar nature (Leitão, Miller et al. 2012). This metric conveys the same information as patch density or mean patch area when the total landscape area is held constant.	Numerical
Mean Patch Area (AREA_MN)	At the class level, AREA_MN is a function of the number of patches in the class and total class area.	Provides a landscape-centric perspective of landscape structure; reflects the average conditions of a pixel chosen at random (McGarigal and Marks 1995).	Hectares

Chapter 4: Results

4.1 Land Accuracy Assessment Results

An error matrix was performed to estimate the number of pixels for the NLCD 2011 urban and non-urban accuracy assessment. The urban land cover class which included aggregated NLCD classes 22-24 yielded a commission error of 100% for the 89 sites visited in the field; whereas including NLCD classes 21 to the aggregated NLC classes 22-24 yielded a 98.88% commission error. The field validation accuracy assessment method for the NLCD 2011 developed land cover classes revealed an overall accuracy of 80.90%, meaning that approximately 81 percent of the urban land cover in 2011 across the CCE was classified correctly. NLCD class 22 (developed, low intensity) had a 78.95% class specific commission error, NLCD class 23 (developed, medium intensity) had an 83.67% class specific commission error, and NLCD class 24 (developed, high intensity) had an 80% class specific commission error (Table 6). NLCD class 21 (developed, open space) differed from classes 22, 23, and 24 by its spatial variability, lack of infrastructure and impervious surface cover, and its low commission error in the field (72.72%).

Table 6. Error matrix of the estimated number of pixels for the NLCD 2011 developed land cover accuracy assessment.

	DO	DL	DM	DH	OL	Column total
DO	8	1	1	0	1	11
DL	1	15	3	0	0	19
DM	1	2	41	5	0	49
DH	0	1	1	8	0	10
OL	0	0	0	0	0	0
Row total	10	19	46	13	1	89

Land cover categories

DO = open space
DL = low intensity
DM = medium intensity
DH = high intensity
OL = other land cover

Overall accuracy = (8+15+41+8+0)/89 = 72/89 = **80.90%**

Omission errors	Comission errors
DO = 8/10 = 80.00%	DO = 8/11 = 72.72%
DL = 15/19 = 78.95%	DL = 15/19 = 78.95%
DM = 41/46 = 89.13%	DM = 41/49 = 83.67%
DH = 8/13 = 61.54%	DH = 8/10 = 80.00%
OL = 0/1 = 00.00%	OL = 0/0 = 00.00%

The accuracy assessment using the Landsat-based imagery validation method revealed varied accuracy in land cover conversion accuracy from 2006-2011. The two conversion types with the highest accuracies were forest to grassland (73.33% commission error) and urban development on agricultural lands (76.67% commission error). The lowest commission error was recorded in the grassland to forest conversion type (16.67%) (Table 7).

I evaluated the overall accuracy, omission error, and commission error for 2006 and 2011 to help explain the results from the land cover change accuracy during 2006-2011. The overall accuracy for all land cover classes in 2006 was 71.62% while the overall accuracy in 2011 was 72.60%. An overall accuracy target of 85% is commonly specified as reliable, and class accuracy set at no less than 70% (Thomlinson, Bolstad et al. 1999). Forested areas had the greatest commission error during 2006 (100%) while urban areas showed the greatest commission error during 2011 (93.33%) (Table 8). Meanwhile, grassland areas had the lowest commission error during both 2006 (38.98%) and during 2011 (50.88%) (Table 9). From this analysis, I confirmed a discrepancy in the grassland land cover class. In 2006, conversion *from* grassland was inaccurately classified as forest 22/59 times (37.29%) and inaccurately classified as agricultural land 14/59 times (23.73%); while in 2011, conversion *to* grassland was inaccurately classified as forest 6/57 times (10.53%) and inaccurately classified as agricultural land 22/57 times (38.60%).

Table 7. Error matrix of the estimated number of pixels for the land cover change accuracy from 2006-2011.

	FG	GF	AG	GA	AU	OC	Column total
FG	22	0	0	0	0	8	30
GF	0	5	0	0	0	25	30
AG	0	0	6	0	0	24	30
GA	0	0	0	13	0	17	30
AU	0	0	0	0	23	7	30
OC	0	0	0	0	0	0	0
Row total	22	5	6	13	23	81	150

Land cover change categories

FG = forest to grassland
GF = grassland to forest
AG = agriculture to grassland
GA = grassland to agriculture
AU = agriculture to urban
OC = other change

Overall accuracy = (22+5+6+13+23+0)/150 = 70/150 = **46.67%**

Omission errors	Comission errors
FG = 22/22 = 100.00%	FG = 22/30 = 73.33%
GF = 5/5 = 100.00%	GF = 5/30 = 16.67%
AG = 6/6 = 100.00%	AG = 6/30 = 20.00%
GA = 13/13 = 100.00%	GA = 13/30 = 43.33%
AU = 23/23 = 100.00%	AU = 23/30 = 76.67%

Table 8. Error matrix of the estimated number of pixels for NLCD 2006.

	F	G	A	U	Column total
F	30	0	0	0	30
G	22	23	14	0	59
A	1	5	53	0	59
U	0	0	0	0	0
Row total	53	28	67	0	148

Land cover categories

F = forest
G = grassland
A = agriculture
U = urban

Overall accuracy = (30+23+53+0)/148 = 106/148 = **71.62%**

Omission errors

F = 30/53 = 56.60%
G = 23/28 = 82.14%
A = 53/67 = 79.10%
U = 0/0 = 00.00%

Comission errors

F = 30/30 = 100.00%
G = 23/59 = 38.98%
A = 53/59 = 89.83%
U = 0/0 = 00.00%

*The urban land cover class is shown with limited accuracy assessment because 1) land cover change from urban to another class is not a land cover conversion type of interest to this study, and 2) land cover change from urban land to another land cover class was not present.

Table 9. Error matrix of the estimated number of pixels for NLCD 2011.

	F	G	A	U	Column total
F	22	7	0	0	29
G	6	29	22	0	57
A	0	3	27	0	30
U	0	1	1	28	30
Row total	28	40	50	28	146

Land cover categories

F = forest
G = grassland
A = agriculture
U = urban

Overall accuracy = (22+29+27+28)/146 = 106/146 = **72.60%**

Omission errors	Comission errors
F = 22/28 = 78.57%	F = 22/29 = 75.86%
G = 29/40 = 72.50%	G = 29/57 = 50.88%
A = 27/50 = 54.00%	A = 27/30 = 90.00%
U = 28/28 = 100.00%	U = 28/30 = 93.33%

4.2 Spatio-temporal patterns of LULCC

To understand the temporal changes of LULCC, I analyzed land cover change for three time periods (1992-2001, 2001-2006, and 2006-2011) from 1992-2011. Again, using the four aggregated land cover classes (open space, forest, urban, and agriculture), the most common land cover conversion from 1992-2001 was forest to open space (18,594.00 km^2), followed by open space to urban (8,613.00 km^2), and agriculture to urban development (7,409.70 km^2) (Table 10). The dominant land cover conversion from 2001-2006 was agriculture to urban (3,045.60 km^2), followed by agriculture to open space (2,492.10 km^2), and agriculture to forest (2,112.30 km^2) (Table 11). The dominant land cover conversion from 2006-2011 was urban development on open space (1,188.00 km^2), followed by agriculture to urban (1,018.80 km^2), and agriculture to forest (2,112.30 km^2) (Table 12).

Table 10. Land cover change in the CCE from 1992-2001.

1992	**2001**	**Total Area of Change (km^2)**
Forest	Open space	18,594.00
Open space	Urban	8,613.00
Agriculture	Urban	7,409.70
Forest	Urban	4,713.30

*This table reflects all land cover conversion *to* and *from* the aggregated land cover classes (open space, forest, urban, and agriculture). Of the aggregated land cover classes, four types of change were present across the study area from 1992-2001.

Table 11. Land cover change in the CCE from 2001-2006.

2001	**2006**	**Total Area of Change (km^2)**
Agriculture	Urban	3,045.60
Agriculture	Open space	2,492.10
Agriculture	Forest	2,112.30
Forest	Agriculture	1,751.40
Open space	Urban	1,181.70
Forest	Open space	383.40
Forest	Urban	172.80

*This table reflects all land cover conversion *to* and *from* the aggregated land cover classes. Seven types of change were present across the study area from 2001-2006.

Table 12. Land cover change in the CCE from 2006-2011.

2006	**2011**	**Total Area of Change (km^2)**
Open space	Urban	1188.00
Agriculture	Urban	1018.80
Agriculture	Forest	863.10
Agriculture	Open space	692.10
Forest	Agriculture	539.10
Forest	Open space	183.60
Forest	Urban	49.50

*This table reflects all land cover conversion *to* and *from* the aggregated land cover classes. Seven types of change were present across the study area from 2006-2011.

From 2001 to 2006 urban development occurred predominantly on agricultural lands (Table 11). From 2006-2011, urban development occurred predominantly on open space lands (Table 12). The shift from urban development occurring predominantly on agricultural areas from 2001-2006, to open space areas from 2006-2011 could mean that agricultural lands weren't necessarily occupying central locations within and near urban areas by 2006. This is also reflected by the fact that land cover change from agriculture to urban development decreased by 66.57% in the CCE from 2001-2011.

From 1992 to 2001 open space development occurred predominantly on forest lands (Table 10). Between 1992-2001 and 2001-2011, open space development underwent a shift from occurring predominantly on forest areas to occurring predominantly on agriculture areas. Additionally, land cover change from open space to urban development increased by 0.53% in the CCE from 2001-2011. Yet, open space development did not see a shift comparable to urban development. Open space development continued to occur predominantly on agriculture lands from 2001-2006 (Table 11) and from 2006-2011 (Table 12), implying a general succession of more intensive land use across the CCE from 1992-2011.

To better characterize the spatial patterns of LULCC in the CCE, I used on the area of Kalispell, Montana to show the spatio-temporal land cover changes of urban development on open space (Figure 5), open space development on rural land (Figure 6), and urban development on rural land (Figure 7) between 1992-2011. Kalispell, Montana is the largest city and commercial center of northwest Montana (U.S. Census) and thus a suitable site to show the spatial trajectory of LULCC. This visualization aided in the comparison between urban development on rural land versus successional urban development on urban land. From the results, during 2001-2006 both urban and open space development on rural area was more centrally located than during 1992-2001. Yet, urban development on rural land was more centralized from 1992-2011 than urban development on open space from 1992-2011.

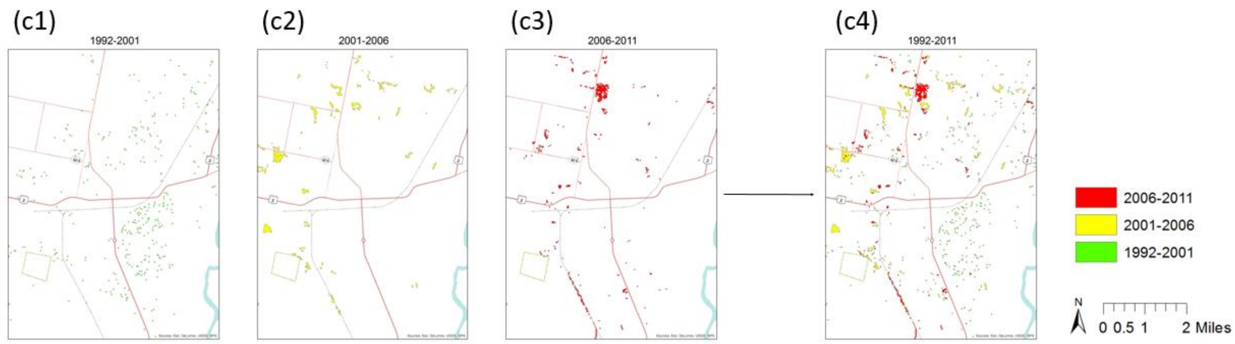

Figure 5. (c) Urban growth on open space in Kalispell, Montana from 1992 to 2011: (c1) Open to urban land cover change from 1992-2001; (c2) Open to urban land cover change from 2001-2006; (c3) Open to urban land cover change from 2006-2011; (c4) Composite image of the open to urban land cover change from 1992-2011. Open space that was converted to urban from 1992 to 2001, from 2001 to 2006, and from 2006 to 2011 are highlighted in green, yellow and red, respectively.

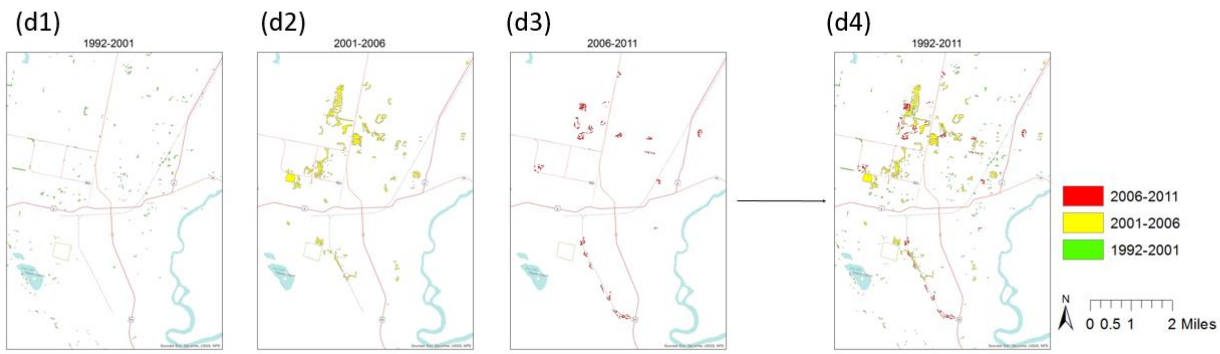

Figure 6. (d) Open space growth on rural land in Kalispell, Montana from 1992 to 2011: (d1) Rural to open space land cover change from 1992-2001; (d2) Rural to open space land cover change from 2001-2006; (d3) Rural to open space land cover change from 2006-2011; (d4) Composite image of the rural to open space land cover change from 1992-2011. Rural areas that were converted to open space from 1992 to 2001, from 2001 to 2006, and from 2006 to 2011 are highlighted in green, yellow and red, respectively. Rural land cover is composed of agriculture and forest land cover classes, however for this analysis rural land cover was only forest areas because of the data available for this conversion.

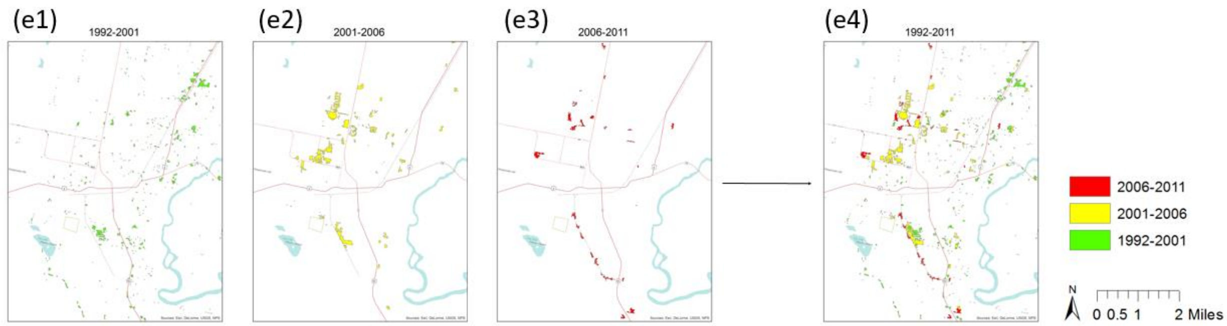

Figure 7. (e) Urban growth on rural land in Kalispell, Montana from 1992 to 2011: (e1) Rural land to urban land cover change from 1992-2001; (e2) Rural land to urban land cover change from 2001-2006; (e3) Rural land to urban land cover change from 2006-2011; (e4) Composite image of the rural to urban land cover change from 1992-2011. Rural land cover (agriculture and forest) that was converted to urban from 1992 to 2001, from 2001 to 2006, and from 2006 to 2011 are highlighted in green, yellow and red, respectively.

4.3 Landscape Metric Analysis Results

4.3.1. CCE landscape metrics

The CCE landscape is dominated by a changing forest land cover matrix (Figure 8). Analysis of the aggregated land cover classes indicated that from 1992-2011, open space and urban land cover increased, while agriculture and forest land cover decreased. The greatest land cover change in the CCE from 1992-2011 occurred on forested land, which experienced a decrease from 24,639,911.11 hectares, or 52.15% of the entire CCE in 1992, to 22,819,907.77 hectares, or 48.35% of the CCE landscape in 2011 (Table 13). The largest forest patch occupied 44.53% of the CCE landscape in 1992 to 33.31% of the CCE landscape in 2011 (Table 13). Meanwhile, the number of agriculture, forest, and open space patches decreased from 1992-2001, and increased from 2001-2011; while the number of urban patches increased during the entire study period, from 1992-2011 (Figure 9). Finally, results indicate that across the entire CCE, the mean patch area of the agriculture class increased from 1992-2001, but decreased from 2001-2011; while the mean patch area of forested areas, which had the largest average patch size of any of the aggregated land cover classes, increased from 109.92 hectares in 1992 to 114.05 hectares in 2001, but decreased to 97.93 hectares from 2001-2006, and further decreased to 88.51 hectares from 2006-2011 (Table 13).

Figure 8. Class area change across the entire CCE from 1992-2011.

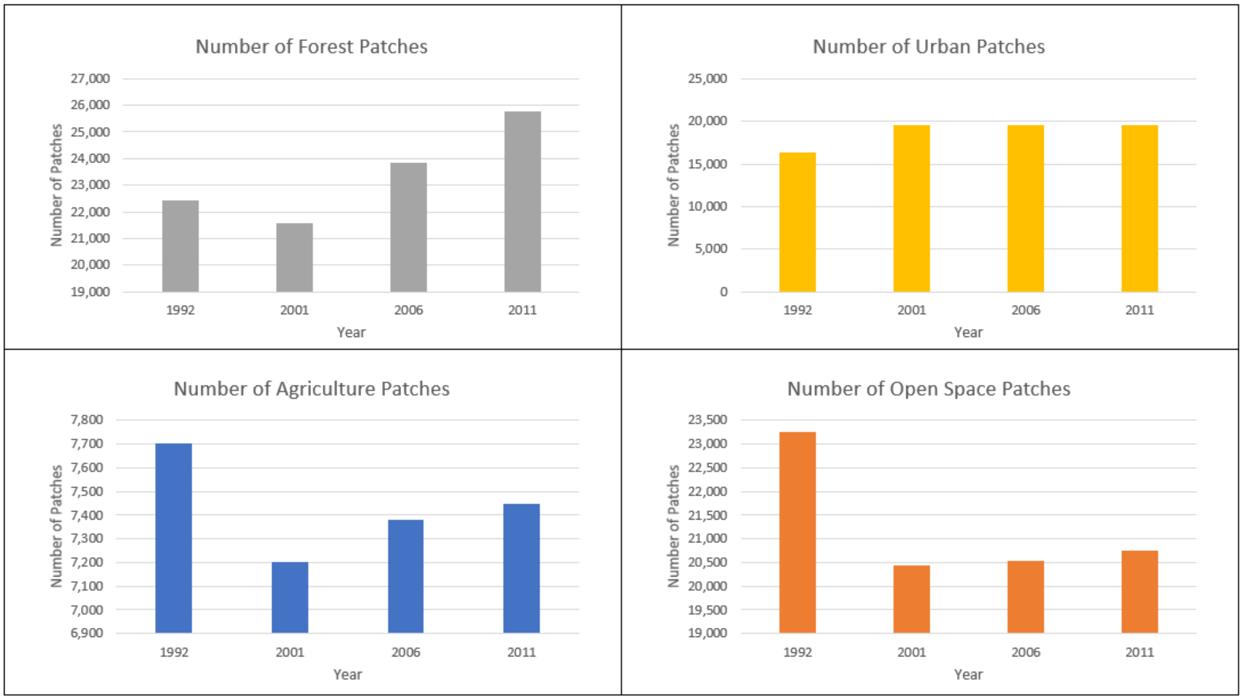

Figure 9. Change in the number of patches across the entire CCE from 1992-2011.

Table 13. Fragstats metric results for the CCE study site.

	1992	**2001**	**2006**	**2011**
Class Area (hectares)				
O	36673.56	37671.66	37948.68	38076.66
F	2463991.11	2461661.91	2333022.30	2281990.77
U	15512.49	18164.88	18688.59	19006.47
A	386840.43	386099.55	382501.08	380732.13
Largest Patch Index (%)				
O	0.01	0.01	0.01	0.01
F	44.53	36.79	34.16	33.31
U	0.07	0.08	0.09	0.09
A	0.40	0.40	0.40	0.40
Number of Patches				
O	23247.00	20448.00	20533.00	20755.00
F	22416.00	21585.00	23823.00	25781.00
U	16333.00	19509.00	19541.00	19609.00
A	7704.00	7201.00	7382.00	7445.00
Mean Patch Area (hectares)				
O	1.58	1.84	1.85	1.83
F	109.92	114.05	97.93	88.51
U	0.95	0.93	0.96	0.97
A	50.21	53.62	51.82	51.14

4.3.2 Case study landscape metrics

KALISPELL

The central Kalispell case study site is composed predominantly of urban and open space land cover (Figure 10). From 1992-2011 analysis revealed that the greatest percent of land cover change in Kalispell occurred amongst the urban land cover class, which experienced an increase from 760 hectares, or 71.87% of the total area in 1992, to 790.38 hectares, or 74.65% of the total study site area by 2011 (Table 14). The largest patch of the urban land cover increased between 1992 and 2011 (Table 14). Meanwhile, the number of urban land cover patches fluctuated, but remained relatively constant 1992-2011 (Table 14). The number of forest and agriculture land cover patches decreased from 1992-2011 while the number of open space patches decreased from 1992-2006 and then increased from 2006-2011 (Table 14). The mean patch area was greatest of the urban land cover class which decreased from 29.27 hectares in 1992 to 26.20 hectares of the Kalispell study area in 2006 (Figure 11). Also, the mean patch area of the open space land cover class decreased from 1992 to 2011 in the central, urban Kalispell study site (Table 14).

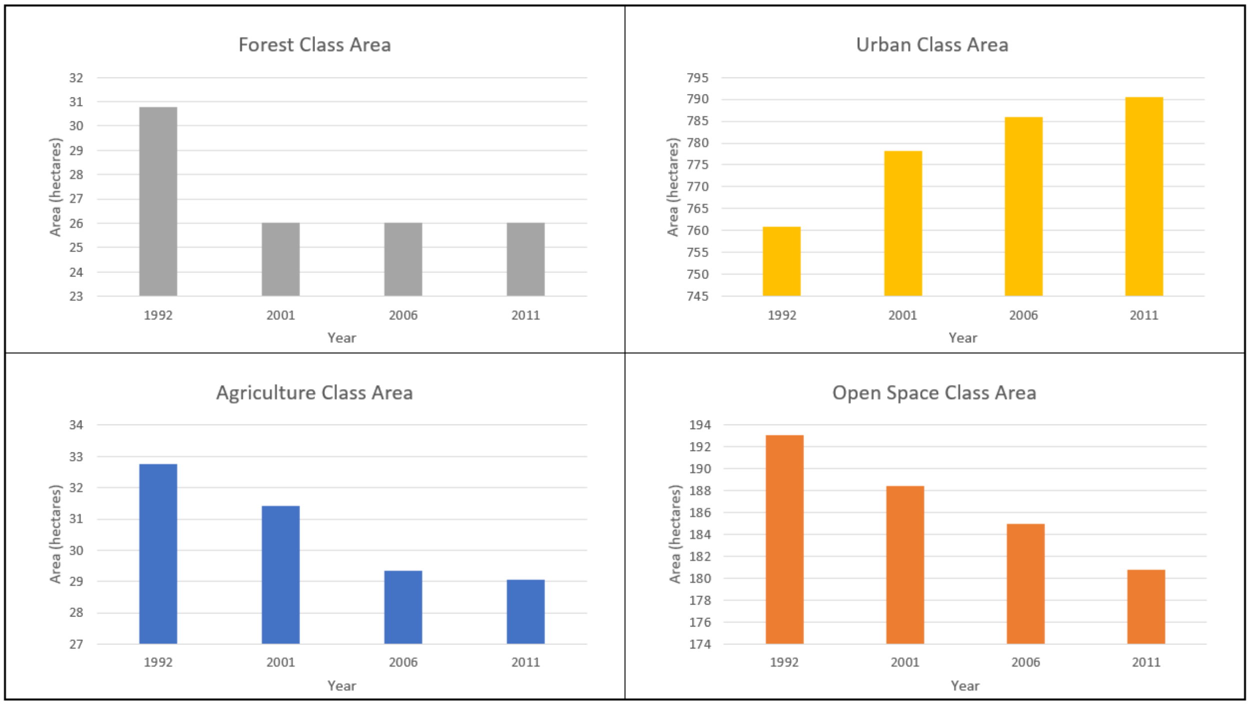

Figure 10. Class area change in the Kalispell study site from 1992-2011.

Figure 11. Mean patch area change in the Kalispell study site from 1992-2011.

Table 14. Fragstats metric results for the Kalispell study site.

	1992	**2001**	**2006**	**2011**
Class Area (hectares)				
O	193.05	188.46	184.95	180.81
F	30.78	26.01	26.01	26.01
U	760.95	778.14	785.97	790.38
A	32.76	31.41	29.34	29.07
Largest Patch Index (%)				
O	5.14	5.18	5.18	5.18
F	1.48	1.44	1.44	1.44
U	70.84	72.35	73.12	73.54
A	0.94	0.94	0.88	0.88
Number of Patches				
O	199.00	198.00	194.00	203.00
F	21.00	16.00	16.00	16.00
U	26.00	32.00	30.00	30.00
A	15.00	13.00	12.00	12.00
Mean Patch Area (hectares)				
O	0.97	0.95	0.95	0.89
F	1.47	1.63	1.63	1.63
U	29.27	24.32	26.20	26.35
A	2.18	2.42	2.45	2.42

THE GREATER KALISPELL AREA

To characterize low density residential development and rural land cover conversion in the Kalispell area, I conducted an additional analysis for the surrounding area of Kalispell. This analysis was conducted separately from the central, urban Kalispell area. The area surrounding Kalispell is composed mostly of agriculture land cover (Figure 12), yet some of the most notable change occurred on forest and urban land cover (Table DEF). The greatest percent of land cover change from 1992-2011 occurred on agriculture land cover, which saw a decrease from 5,177.88 hectares, or 38.12% of the total area in 1992, to 4,660.47 hectares, or 34.31% of the surrounding Kalispell area by 2011 (Table 15). The largest patch of the agriculture land cover class decreased from 9.27% of the total study site area in 1992 to 7.31% by 2011 (Table 15). Meanwhile, the number of forest and agriculture land cover patches decreased from 1992-2001, and then remained relatively constant from 2001-2011 (Table 15). The mean patch area was greatest for the agriculture land cover class during the entire time period, and increased from 1992-2001 and then decreased from 2001-2011, as did the forest land cover class (Figure 13). The mean patch area for the urban land cover class increased from 1992-2011, while the mean patch area for the open space land cover class increased from 1992-2006 and then decreased from 2006-2011 (Figure 13). Additionally, notable land cover change in the greater Kalispell area comprised change from agriculture to open space land cover, change from agriculture to urban land cover, and overall expansion of open space and urban land cover, especially at the agriculture-urban periphery (Figure 14).

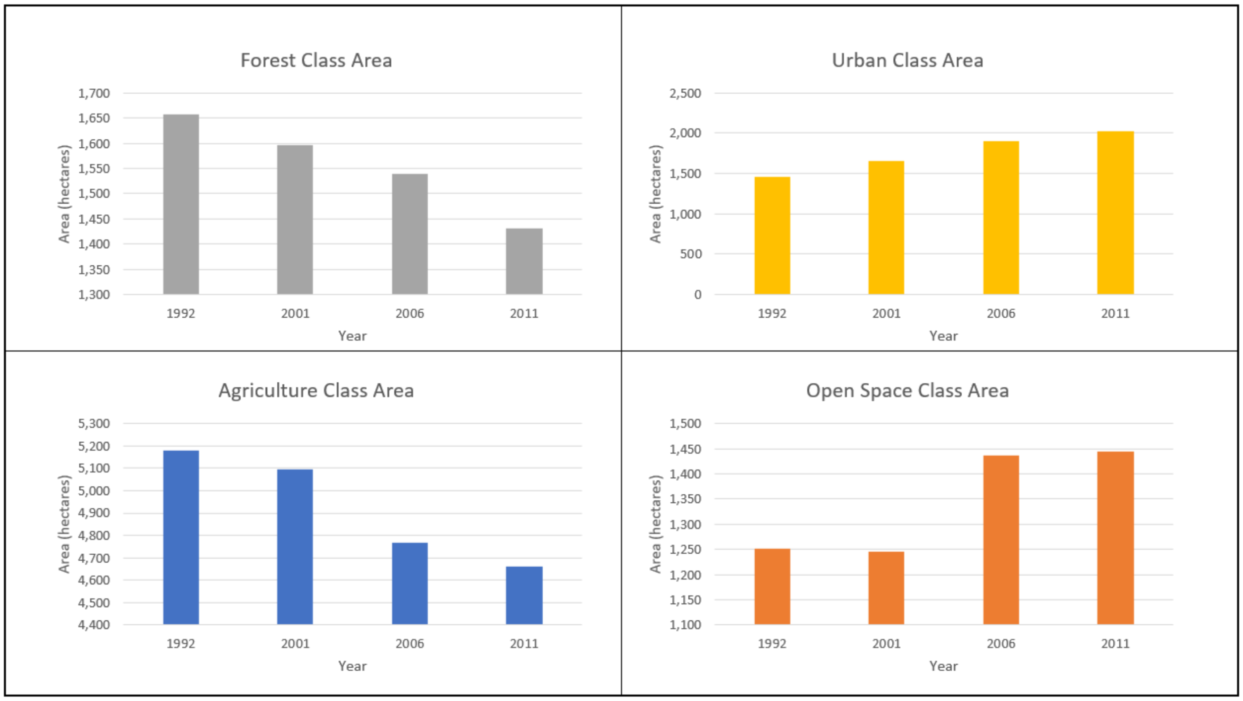

Figure 12. Class area change in the in the area surrounding Kalispell from 1992-2011.

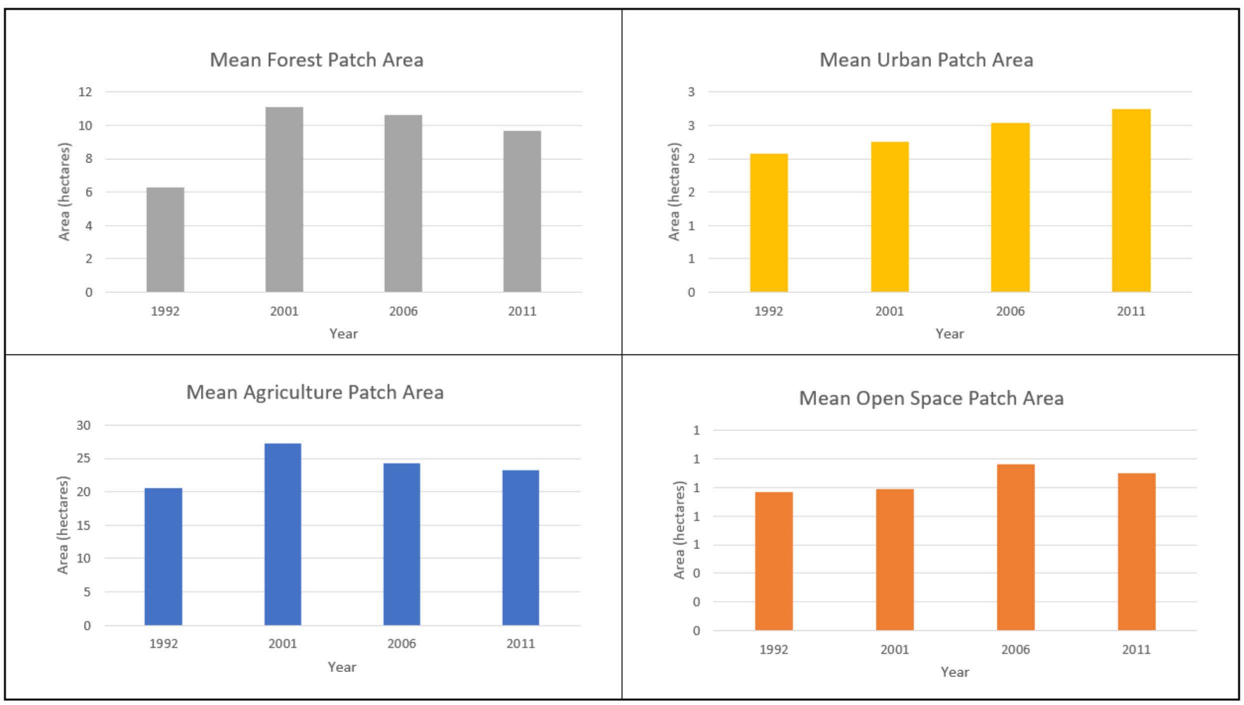

Figure 21. Mean patch area change in the area surrounding Kalispell from 1992-2011.

Table 15. Fragstats metric results for the area surrounding Kalispell.

	1992	2001	2006	2011
Class Area (hectares)				
O	1252.17	1245.60	1437.12	1444.14
F	1657.71	1595.61	1539.00	1431.90
U	1463.49	1658.52	1899.81	2025.72
A	5177.88	5096.70	4769.37	4660.47
Largest Patch Index (%)				
O	0.61	0.60	0.59	0.58
F	5.15	5.13	4.61	3.87
U	3.43	4.97	5.20	5.29
A	9.27	9.25	7.86	7.31
Number of Patches				
O	1296.00	1255.00	1235.00	1310.00
F	263.00	187.00	145.00	148.00
U	705.00	737.00	750.00	738.00
A	251.00	187.00	196.00	201.00
Mean Patch Area (hectares)				
O	0.97	0.99	1.16	1.10
F	6.30	11.08	10.61	9.68
U	2.08	2.25	2.53	2.74
A	20.63	27.26	24.33	23.19

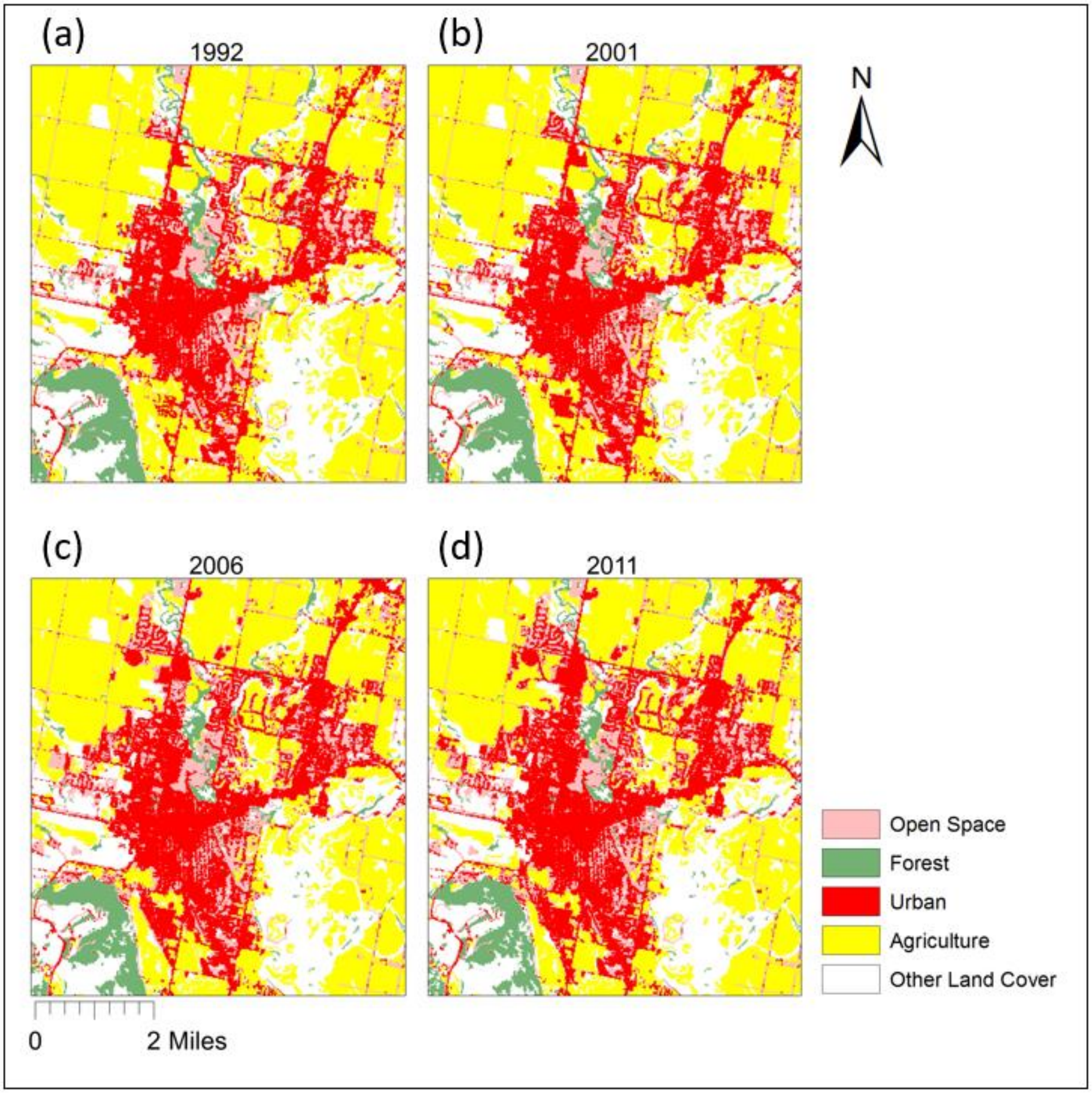

Figure 14. Land cover in Kalispell, Montana from 1992 to 2011 at a 1:50,000 scale where each pixel represents 30x30 m data: (a) Kalispell, Montana in 1992; (b) Kalispell, Montana in 2001; (c) Kalispell, Montana in 2006; and (d) Kalispell, Montana in 2011.

WHITEFISH

The central Whitefish study site is composed mostly of urban land cover (Figure 15). The greatest percent of land cover change from 1992-2011 occurred on urban land, which saw close to a 5% increase from 1992 to 2011 (Table 16). Urban land cover occupied the largest patch index, and increased from 26.98% in 1992 to 29.83% by 2011 (Table 16). The number of urban patches increased most from 1992-2001 while the number of open space and agriculture patches remained relatively constant from 1992-2011 (Table 16). The mean patch area was greatest for the agriculture land cover class during the entire time period, yet increased from 1992-2001 and then decreased from 2001-2011, as did the forest land cover class (Figure 16). Finally, the mean patch area for the open space and urban land cover classes increased from 1992-2001, while the mean patch area for forest and agriculture areas increased from 1992-2006 and then decreased from 2006-2011 (Figure 16 and Table 16).

Figure 15. Class area change in the Whitefish study site from 1992-2011.

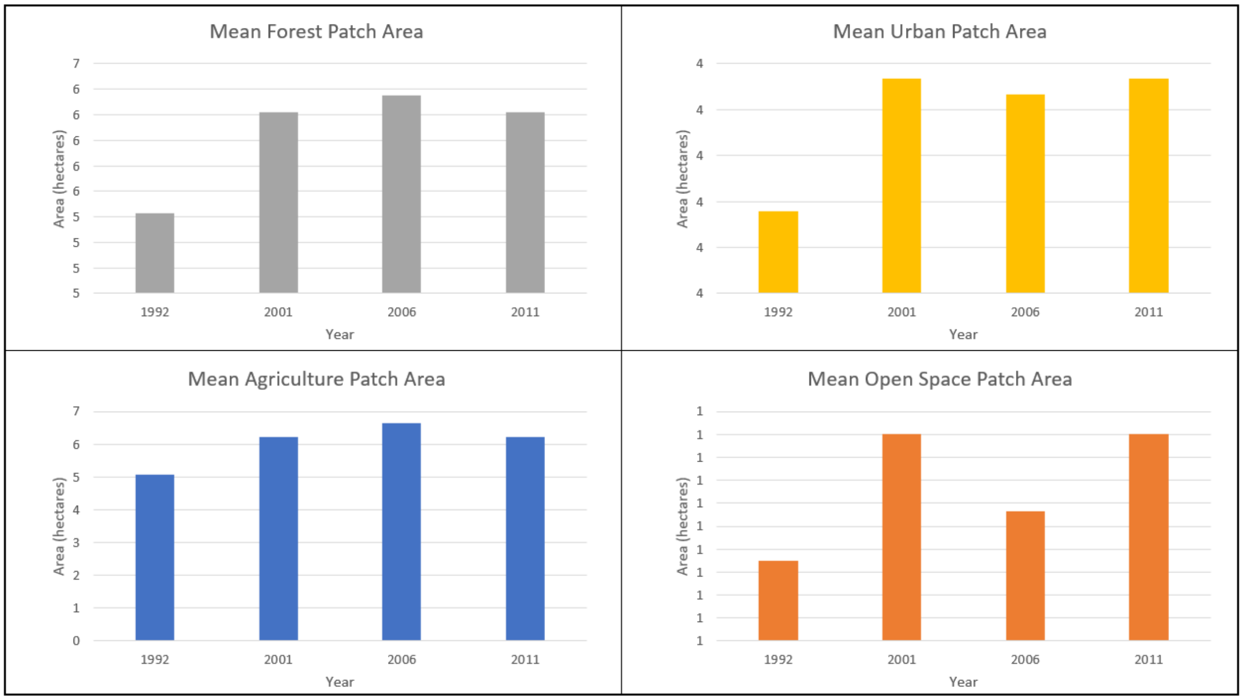

Figure 16. Mean patch area change in the Whitefish study site from 1992-2011.

Table 16. Fragstats metric results for the Whitefish study site.

	1992	**2001**	**2006**	**2011**
Class Area (hectares)				
O	161.01	162.72	164.97	167.40
F	233.37	220.86	216.00	211.59
U	302.04	318.87	339.03	344.34
A	198.00	194.76	179.10	174.06
Largest Patch Index (%)				
O	3.44	4.12	3.14	3.28
F	9.95	9.23	9.23	9.11
U	26.98	27.93	29.73	29.83
A	6.12	6.11	6.05	5.62
Number of Patches				
O	184.00	178.00	184.00	180.00
F	43.00	35.00	34.00	34.00
U	83.00	90.00	90.00	91.00
A	39.00	31.00	27.00	28.00
Mean Patch Area (hectares)				
O	0.88	0.93	0.90	0.93
F	5.43	6.22	6.35	6.22
U	3.64	3.78	3.77	3.78
A	5.08	6.22	6.63	6.22

THE GREATER WHITEFISH AREA

The surrounding Whitefish study site is set in a forest land cover matrix (Figure 17). From 1992-2011, the greatest percent of land cover change occurred on forested land, which saw a decrease from 59.94% of the total surrounding Whitefish area in 1992, to 55.28% of the total surrounding Whitefish area by 2011 (Table 17). Also, the largest patch index for the urban land cover class increased from 0.42% to 1.14% of the study area from 1992-2011 (Table 17). Additionally, the number of patches for the open space land cover class decreased greatly from 1992-2001 but then increased from 2001-2011, while the number of urban patches increased greatly from 1992-2001 and then remained relatively unchanged from 2001-2011 (Table 17). Additionally, the mean patch area for urban areas increased from 1992-2011, while the open space, forest, and agriculture areas all increased from 1992-2001, but then decreased or stayed relatively constant (Table 17).

Figure 17. Class area change in the surrounding Whitefish area from 1992-2011.

Table 17. Fragstats metric results for the surrounding Whitefish area.

	1992	**2001**	**2006**	**2011**
Class Area (hectares)				
O	935.28	1095.12	1110.33	1102.68
F	11931.21	11651.49	11427.21	11003.31
U	467.82	616.32	685.35	696.33
A	4190.76	4172.40	4119.12	4100.76
Largest Patch Index (%)				
O	0.25	0.35	0.33	0.32
F	20.89	20.57	20.31	19.02
U	0.42	0.78	1.10	1.14
A	8.92	8.90	8.85	8.83
Number of Patches				
O	953.00	779.00	819.00	828.00
F	262.00	246.00	253.00	259.00
U	609.00	746.00	745.00	743.00
A	157.00	148.00	153.00	150.00
Mean Patch Area (hectares)				
O	0.98	1.41	1.36	1.33
F	45.54	47.36	45.17	42.48
U	0.77	0.83	0.92	0.94
A	26.69	28.19	26.92	27.34

WEST GLACIER

West Glacier is dominated by forest land cover matrix (Figure 18). Despite urban and open space occupying a small amount of the West Glacier landscape, both land cover classes saw an increase, or near constant state, from 1992-2011 (Figure 18). Forested area decreased from 83.28% of the total landscape in 1992 to 78.47% in 2011 (Table 18). The largest patch index for the forest and agriculture classes remained relatively constant and the largest patch index for the urban land cover increased from 0.49% of the landscape in 1992 to 0.67% in 2006 and 2011 (Table 18). Interestingly, the number of urban patches increased greatly from 1992-2001 and then remained relatively constant from 2001-2011, all while the number of patches for agriculture areas decreased from 1992-2011 (Table 18). Furthermore, the open space area dropped overall from 1992 to 2011. Also, the forest, open space, and agriculture mean patch size increased from 1992-2001 but decreased from 2001-2011, while urban mean patch size decreased from 1992-2001 but then stayed relatively constant from 2001-2011 (Figure 19). Overall, urban areas increased from 1992-2011 and that the mean patch size of open space land cover also increased from 1992-2011 timeframe within this study site (Table 18).

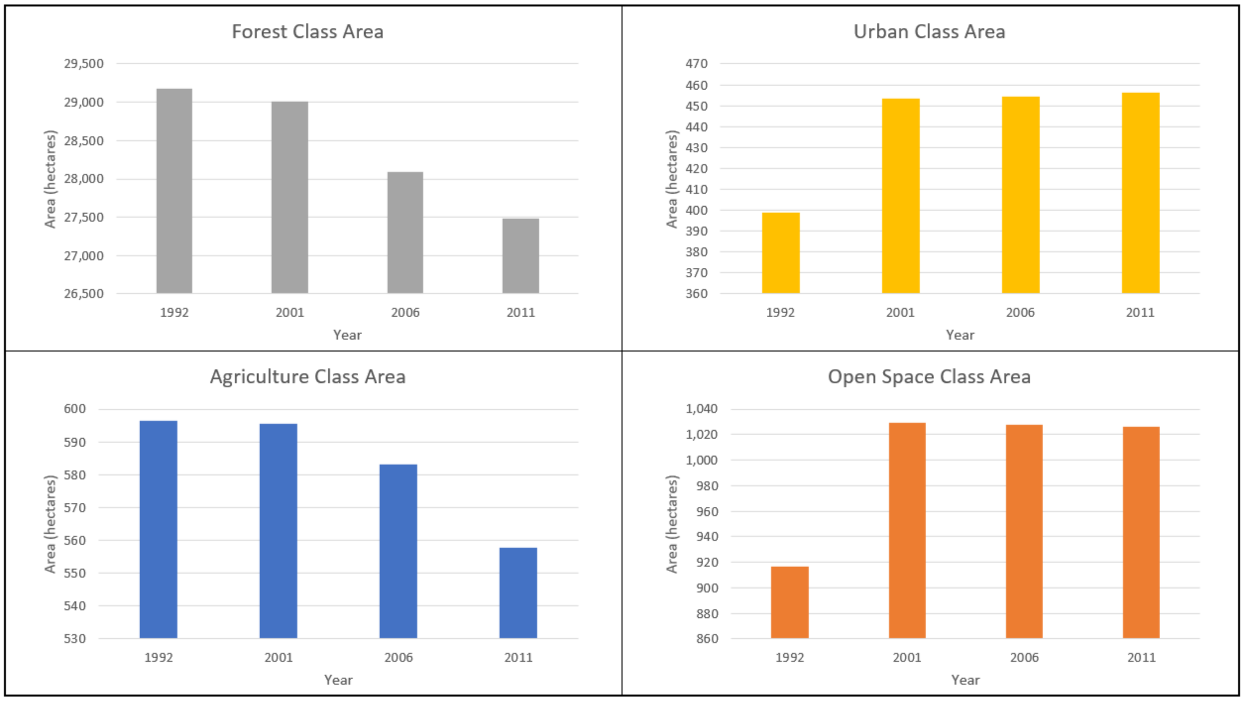

Figure 18. Class area change in the West Glacier study site from 1992-2011.

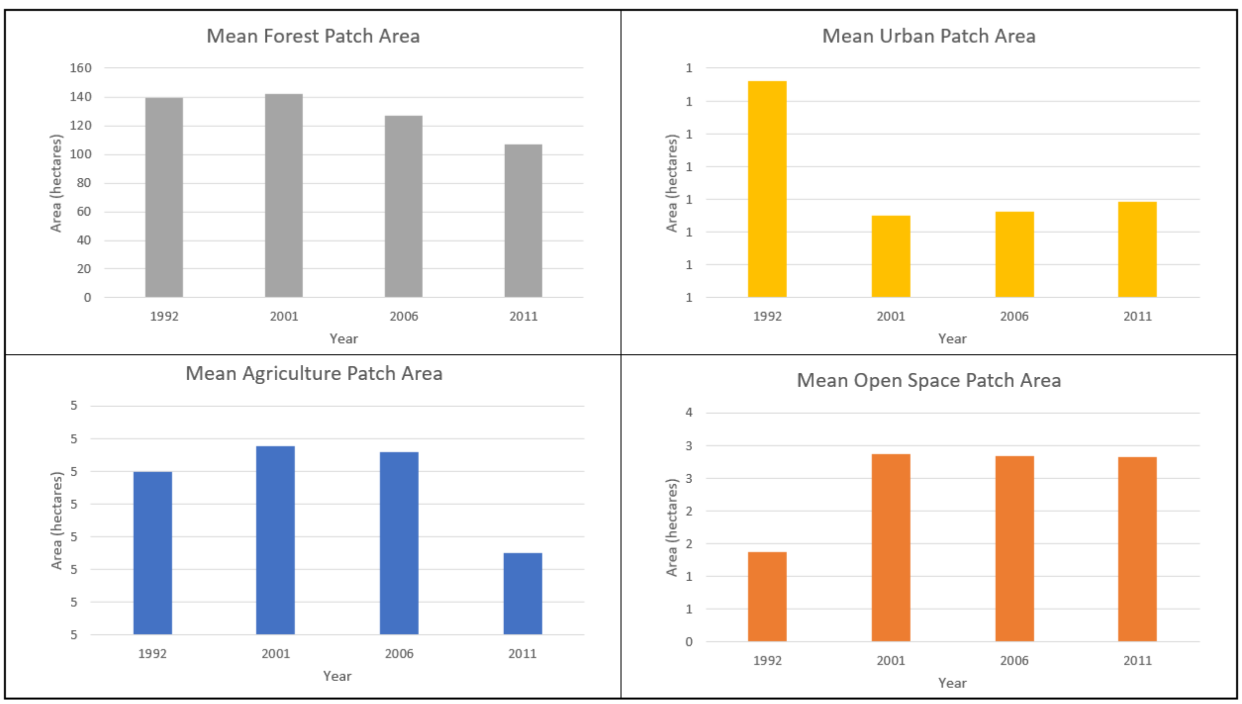

Figure 19. Mean patch area change in the West Glacier study site from 1992-2011.

Table 18. Fragstats metric results for the West Glacier study site.

	1992	**2001**	**2006**	**2011**
Class Area (hectares)				
O	916.56	1028.97	1027.80	1025.91
F	29173.50	29012.13	28084.77	27487.80
U	398.70	453.33	454.50	456.39
A	596.52	595.44	583.02	557.82
Largest Patch Index (%)				
O	0.43	0.54	0.39	0.39
F	54.16	53.96	53.89	53.14
U	0.49	0.66	0.67	0.67
A	0.32	0.32	0.32	0.32
Number of Patches				
O	665.00	358.00	362.00	363.00
F	209.00	204.00	221.00	257.00
U	324.00	442.00	441.00	436.00
A	117.00	115.00	113.00	115.00
Mean Patch Area (hectares)				
O	1.38	2.87	2.84	2.83
F	139.59	142.22	127.08	106.96
U	1.23	1.03	1.03	1.05
A	5.10	5.18	5.16	4.85

POLEBRIDGE

The area of Polebridge, located just outside the northwestern corner of Glacier National Park, is composed mostly of forest land cover. No areas of open space and urban land cover are present in this study area (Figure 20 and Table 20). From 1992-2011, forest and agriculture land cover classes experienced very little change, undergoing an overall decrease in percent total area of 89.78% in 1992 to 84.39% in 2011 (Table 19). Little observed change occurred for the agriculture land cover class' largest patch index. Largest patch index for the forest class decreased from 89.36% in 1992 to 83.74% in 2011 (Table 19). The forest land cover also had an increase in the number of patches from 2001-2011, with the greatest change occurring from 1992-2006 (Table 19). Meanwhile, the average patch size of agriculture land class increased from 1992-2011 while the average patch size of the forest land cover class decreased from 371.46 hectares in 2001 to 239.06 hectares in 2011. The study area experienced an overall decrease in both the agriculture and forest land cover classes, while experiencing an increase in other land cover from 1992-2011 (Table 19).

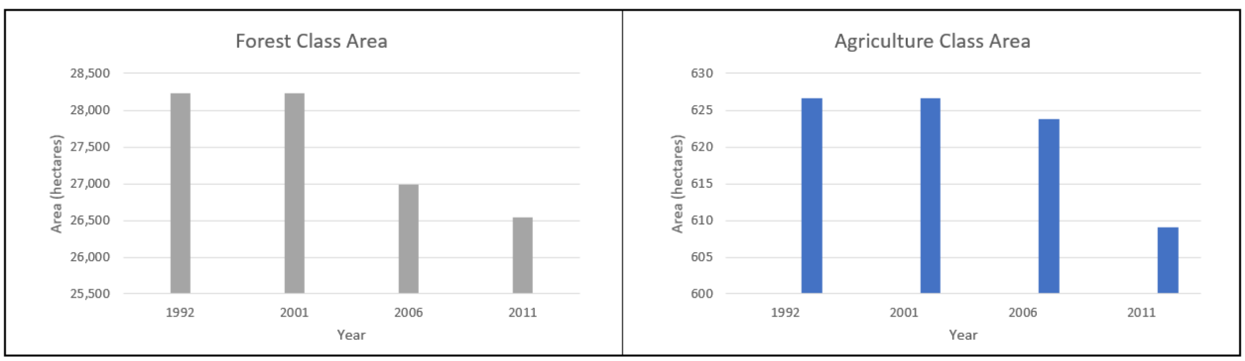

Figure 20. Class area change in the Polebridge study site from 1992-2011.

Table 19. Fragstats metric results for the Polebridge study site.

	1992	2001	2006	2011
Class Area (hectares)				
O	-	-	-	-
F	28230.93	28230.93	26982.18	26536.14
U	-	-	-	-
A	626.67	626.67	623.79	609.12
Percent of Landscape (%)				
O	-	-	-	-
F	89.78	89.78	85.81	84.39
U	-	-	-	-
A	1.99	1.99	1.98	1.94
Largest Patch Index (%)				
O	-	-	-	-
F	89.36	89.36	85.27	83.74
U	-	-	-	-
A	1.08	1.08	1.08	1.06
Number of Patches				
O	-	-	-	-
F	76.00	76.00	101.00	111.00
U	-	-	-	-
A	87.00	87.00	86.00	83.00
Mean Patch Area (hectares)				
O	-	-	-	-
F	371.46	371.46	267.15	239.06
U	-	-	-	-
A	7.20	7.20	7.25	7.34

Table 20. Annual rate of land cover change in the entire CCE study area, Kalispell, Whitefish, West Glacier, and Polebridge study sites.

	1992-2001	2001-2006	2006-2011
CCE			
O	0.29%	0.15%	-0.45%
F	-0.01%	-1.10%	0.33%
U	1.62%	0.56%	-0.09%
A	-0.02%	-0.19%	-0.45%
Kalispell			
O	-0.27%	-0.38%	-0.46%
F	-2.04%	0.00%	0.00%
U	0.25%	0.20%	0.11%
A	-0.48%	-1.41%	-0.19%
Surrounding Kalispell Area			
O	-0.06%	2.67%	0.10%
F	-0.43%	-0.74%	-1.50%
U	1.31%	2.54%	1.24%
A	-0.18%	-1.37%	-0.47%
Whitefish			
O	0.12%	0.27%	0.29%
F	-0.63%	-0.45%	-0.42%
U	0.59%	1.19%	0.31%
A	-0.18%	-1.75%	-0.58%
Surrounding Whitefish Area			
O	1.62%	0.27%	-0.14%
F	-0.27%	-0.39%	-0.77%
U	2.68%	2.01%	0.32%
A	-0.05%	-0.26%	-0.09%
West Glacier			
O	1.21%	-0.02%	-0.04%
F	-0.06%	-0.66%	-0.43%
U	1.34%	0.05%	0.08%
A	-0.02%	-0.43%	-0.90%
Polebridge			
O	-	-	-
F	0.00%	-0.93%	-0.34%
U	-	-	-
A	0.00%	-0.09%	-0.48%

*Values without a symbol represent rate increase, whereas values with (-) represent rate decrease.

Chapter 5: Discussion and Conclusion

5.1 Discussion

5.1.1 Accuracy Assessment

I conducted an accuracy assessment specifically to determine the applicability of NLCD for analyzing land cover change in the CCE, as determined using an accuracy assessment from both field-validated data and from Landsat-derived digital imagery. The main findings from the accuracy assessments can be summarized as follows. First, the usefulness of national scale products for local application varied regionally, temporally, and varied for each land cover type. Second, from the field validation, aggregating each of the developed NLCD classes, except 21 (developed, open space) into one 'urban' class produced 100% accuracy, meaning that the inclusion of NLCD class 22 (developed, low intensity) did not decrease the urban accuracy assessment. Finally, I suggest, based on this work, that national scale *products did not provide* highly accurate land cover change data for local application, *unless* aggregated into broader classes which would offset the discrepancy between like classes.

Previous work by Wickham, Stehman et al. (2013) reported discrepancies between NLCD class 21 (developed, open space), NLCD class 71 (grassland/herbaceous), NLCD class 81 (pasture/hay), and NLCD class 82 (cultivated crops). As determined by the accuracy assessment it is clear that within the study site, the accuracy of NLCD data for local application also varies with land cover class and time period. For example, grassland areas had the lowest commission error during both 2006 (38.98%) and during 2011 (50.88%) (Table 9). Yet, aggregating some of the land cover classes to the Anderson Level I classification scheme greatly improved accuracy of those classes. For example, forested areas had the greatest commission error during 2006

(100%) while urban areas showed the greatest commission error during 2011 (93.33%) (Table 9).

Field validation of the accuracy assessment was performed for the non-aggregated, developed NLCD classes (21, 22, 23 and 24). As mentioned, excluding NLCD class 21 (developed, open space) from NLCD classes 22, 23, and 24 allowed us to better characterize successional stages of urban development by observing the lowest form of development as a separate entity without compromising the accuracy of aggregated urban land cover class. Much of the literature surrounding urban class aggregation found that commission error increased as the level of urbanization increased (Wickham, Stehman et al. 2013), and thus aggregation of NLCD classes 23 and 24 was found to be common in the accuracy assessment literature.

Finally, evaluating the accuracy of land cover data is often not accounted for in local application, nor is the assessment of land cover data through a land cover change method. However, using the two-part accuracy assessment results, I conclude that the NLCD products are relatively accurate when used to assess the spatial and temporal patterns of LULCC in the CCE, and possibly the Intermountain region.

The urban accuracy assessment performed with the GPS revealed that NLCD class 21 (developed, open space) had the lowest class-specific accuracy (72.72%) compared to NLCD class 22 (developed, low intensity) which had a class-specific accuracy of 78.95%, class 23 (developed, medium intensity) which had an overall accuracy of 83.67%, and class 24 (developed, high intensity) which had a class-specific accuracy of 80% (Table 9). Those pixels that were correctly identified as developed, open space occupied areas that were composed of lawn-like grasses typically located on the periphery of urban areas, parking areas, or major roadways. These areas were often situated adjacent to existing neighborhoods, RV parks, shopping centers, warehouses and corporate buildings, and newly developed golf course communities. The inaccuracies of the developed, open space land cover class could also be attributed to the fact that ground truth analysis was performed during the months of June and July of 2015. In fact, during the urban accuracy assessment the findings suggest that many of the inaccuracies were attributed to underestimated degrees of which the land had been developed. Often, the underestimation of urban development was most prevalent in areas of new construction, and in new-build neighborhoods set in a predominantly mixed forest settings. Accuracy assessment performed at the class-level had not previously been performed strictly for the CCE area, though Wickham et al. (2013) had employed regional stratification at the class-level that included the CCE and some of the Intermountain West region. They found that single date accuracies translated into high overall accuracies across all regions (98% in the Intermountain West region) for the binary change–no change classification (Wickham, Stehman et al. 2013).

The aggregated land cover classes (open space, forest, urban, grassland, and agriculture) used for the Google Earth accuracy assessment were selected for the purpose of investigating the accuracy of human-influenced land cover change. In alignment with this work, the findings suggest frequent error between forest and grassland land cover classes during 2006 (Table 8), and forest, grassland, and agriculture land cover classes during 2011 (Table 9) based on the aggregated class-level accuracy assessment.

The accuracy assessment from the field validation revealed that urban areas were typically misclassified in the degree to which they were developed, meaning that areas were often more developed than predicted in the NLCD change map. This discrepancy may be due to the fact that the reference data was collected in 2015 while the media used to label the map was from 2011.

Given the rapid rate of land cover change occurring in the region, I acknowledge that this time lapse increased the likelihood that differences might have occurred due to changes in land cover rather than misclassification (Bossler, McMaster et al. 2010).

The urban accuracy assessment performed in the field enabled us to rationalize the categories used in the assessment of LULCC in the CCE, and helped verify the usefulness of NLCD data for this research. I combined classes 22 (developed, low intensity), 23 (developed, medium intensity), and 24 (developed, high intensity) into one overarching urban class representative of the Anderson II classification scheme. Once aggregated, these classes had a 100% accuracy in the field and therefore were useful for this study. We found that urban development can be more broadly aggregated than the previous literature cites (Wickham, Stehman et al. 2013). In the CCE, and possibly other regions in the northern Intermountain West, NLCD classes 22-24 can be aggregated to represent urban areas without compromising land cover accuracy.

The improvement in accuracy from class aggregation indicates that much of the disagreement was not contained within classes in the same hierarchical group. This is due to the fact that urban areas were not typically misclassified, but instead did not accurately report the degree to which the land was developed. A pitfall of using NLCD is that it is most accurate once aggregated to the Anderson Level II classification scheme, therefore class-specific studies could warrant inaccurate analysis.

Accurate detection of change relies on accurate classification at both dates. The individual class accuracy assessment accounted for three classes in 2006 (forest, grassland, and agriculture) and four classes in 2011 (forest, grassland, agriculture, and urban). Urban development was responsible for the additional land cover class in the 2011 class specific accuracy assessment. Accounting for the accuracy of the urban land cover class may have resulted in a higher overall accuracy for 2011 than in 2006.

I conclude that national scale *products did not provide* highly accurate land cover change data for local application, unless aggregated into broader classes which would offset the discrepancy between like classes. Additionally, agriculture, specifically cropland, is confined mainly to flat areas (Alfieri, Niyogi et al. 2007), which could lend itself to greater NLCD accuracy. Foody et al. (2002) showed that topographically complex regions can impact the classification of land cover accuracy. It is possible that in the study area, class accuracies were also impacted by topography as well.

Additionally, the NLCD 1992 land cover data, which displays a different classification scheme, did not accurately display forest land cover and showed inconsistencies even after manual correction. Before applying the smoothing algorithm, the agriculture, forest, and urban land cover classes of the 1992 land cover map were not spatially similar in composition to those of the 2001 land cover class. A discrepancy is apparent in the underestimation of the mean patch area for the forest land cover class throughout the various case study sites. For example, on the surrounding Kalispell landscape, the mean patch area of the forest land cover class nearly doubled from 1992-2001 and then decreased from 2001-2011.

Finally, use of a land cover change map to assess NLCD accuracy is an effective method for evaluating landscapes undergoing rapid land use change. In the Intermountain West, LULCC is rapidly occurring and thus evaluating a change map allowed us to concentrate on those areas undergoing change.

5.1.2 Spatio-temporal Patterns of LULCC from 1992-2011

LULCC in the CCE region from 1992-2011 suggests that urban development became the most common land cover type, and typically occurred predominantly on agricultural land cover and developed, open space land cover. Open space land cover, the lowest degree of urban development, occurred mostly on forest land cover from 1992-2001, then occurred mostly on agriculture land cover from 2001-2006, and then became the most common land cover type for urban development to take over from 2006-2011. Open space land cover, defined by the MRLC, is typically any area composed mostly of vegetation in the form of lawn-like grass near built areas for aesthetic purposes. This shift, a succession of more intense land uses, shows the increasing urbanization of the CCE landscape from 1992-2011 despite the fact that urban development generally decreased from 1992-2011.

Additionally, open space development on forested areas decreased dramatically (by 99.01%) between 1992 and 2011 (Table 13-15). Urban development on forest areas also decreased greatly from 1992-2011 by 98.95% (Table 13-15); however, this type of land cover conversion was far less prevalent in the CCE than urban development on forests from 1992-2011. Furthermore, the succession of forested areas by higher degrees of urban development aligns with the literature suggesting the amenity driven population growth correlates with forest cover, among other physical attributes of the landscape (Rasker and Hansen 2000). Open space development on forested areas may indicate high amenity, second home development. Additionally, urban and open space development on forest land cover, despite continually decreasing from 1992-2011, could have implications for landscape fragmentation and those threats caused by urban areas pressing against some of the few remaining intact lands for large species (McCool, Adams et al. 2006, Prato 2012).

Still, conversion of forested areas to agriculture land, though not present from 1992-2001, decreased between 2001 and 2011. This trend could suggest a general decrease in new agriculture ventures across the region, as well as the replacement of the agriculture-dependent economy by the recreation-driven economy. It could also imply that the occurrence of agriculture taking over forested areas is not a common trend of the study's time period, which could be attributed to the location of intact forests on mountainous terrain, which tends to be unsuitable for modern farming. A likely explanation for decreasing agriculture land cover is that agricultural land is highly sought after by land developers and private land owners. It is known that the threat of urban growth, a parallel trend I have observed, leads to greater fragmentation of other land uses, such as forest and agriculture (Nagendra, Munroe et al. 2004).

5.1.3 Landscape Metric Analysis: Case Studies

Located at the intersection of two major highways in the Flathead Valley, Kalispell, Montana also offers access to three ski resorts, a golf course, the Flathead Lake, a major airport, and Glacier National Park, all accessible by vehicle within a 40 mile commute. In fact, the area is predominantly urban set in an agriculture setting. From 1992-2011 the area immediately surrounding Kalispell saw some of the greatest rates of forest land cover decrease in the CCE, especially from 2006-2011 (Table 20). This area just outside of Kalispell also saw some of the greatest urban land cover increase (2.54% from 2001-2006) compared to the CCE area (0.56% from 2001-2011), and the greatest rate of open space increase (2.67% from 2001-2006) when compared against the CCE (0.15% from 2001-2006) (Table 20).

Whitefish, Montana, located only 15 miles north of Kalispell, saw similar trends of urban and open space land cover growth on its adjacent, periphery sites as Kalispell experienced. The more central, urban area of Whitefish also saw increased rates of urban development (1.19% from 2001-2006) when compared against the CCE (0.56% from 2001-2006); and exhibited similar trends to the Kalispell area in regards to the lands surrounding urban areas which saw above average urban land cover increase from 1992-2011 (Table 20). Yet, the central Kalispell area saw a decrease in open space land cover from 1992-2011 while the central Whitefish area increased, suggesting that Whitefish, in part because it is less developed than Kalispell, is experiencing an overflow of urban development interest from land developers that look to Kalispell for second homes and recreational opportunities.

Historically, the flatter regions in the CCE were heavily used for agriculture (Prato and Fagre 2010). This includes the areas of Whitefish and Kalispell, Montana which today still exhibit characteristics of the agriculture industry as well as agriculture land cover. Only a few decades ago Montana as a whole seemed defund of capitalistic, resort-based ventures (Malone, Roeder et al. 1991). Yet, National forests host a great deal of recreation. In fact, Flathead National Forest was visited 1.4 million times in 2000 (Prato and Fagre 2010). I suggest that rural areas adjacent to urban areas are common places for further urban development. However, this form of development has implications for landscape connectivity and species that use lowland areas to migrate and move between higher elevation areas. Large carnivorous species often use valley bottoms for food sources. For example, the endangered wolf (*Canis lupus*) killed deer in valley bottoms along wolf travel routes more than on any other landscape type (Kunkel and Pletscher 2001).

From my analysis, I suggest that the area of Whitefish has succumbed to urban development due in part to more recent amenity migration than the nearby Kalispell area. Whitefish is surrounded by natural and cultural amenities, as well as recreational opportunities such as forests, mountains, and ski resorts, that draw thousands of tourists, and second home buyers each year (Prato and Fagre 2010). The number of open space patches in the surrounding Kalispell area generally decreased from 1992-2006 but increased from 2006-2011 possibly indicating greater urban sprawl and urban expansion during the 2006-2011 time frame, at least in areas where urban development was quite common prior to the study period.

Meanwhile, West Glacier, Montana, also located in Flathead County, is a gateway community, or access point to Glacier National Park which is located at the west entrance to the Going-to-the-Sun Road. This road remains a popular east-west trending highway traversing the Lewis Range in Glacier National Park. This area exhibits many of the characteristics of an amenity town; for example, West Glacier is set in a wilderness context, dominated by forest land cover matrix, and has experienced an increase in forest fragmentation, characterized by a growing increase in the number of forest patches from 2001-2011 (Figure 18). Unlike Whitefish and Kalispell, West Glacier does not show the same rates of land cover change, but does show many of the same land cover change trends, such as decreasing forest and agriculture land cover, and increasing urban development.

Forest land cover change in the West Glacier study site might suggest development to accommodate park visitors. Additionally, West Glacier saw no change in the average patch size of urban and open space land cover from 2001-2011 which might reveal two phenomena. First, urban development directly adjacent to Glacier National Park might not be as common as I had predicted because of possible development restrictions and human preference for access to various recreational outlets (ATV trails, biking trails, shopping districts, golf courses, etc.). Second, because of the lack of urban development from 2001-2011, I suggest that NLCD does not accurately reflect low density urban development in the CCE, especially in heavily forested areas, or areas with topographic complexity that might not enable accurate detection of residential or impervious cover. Additionally, NLCD and land cover maps cannot always accurately explain land use.

The study site of West Glacier experienced an increase in the number of forest patches while experiencing a decrease in the average forest patch size from 1992-2011. One exception to this trend was found from 1992-2001, though I suspect this error to be attributed to the land cover dataset. Regardless, these characteristics could indicate decreased landscape functionality and connectivity due to human activities, mountain bark beetle infestation, or forest fires. Decreased landscape connectivity can affect the functionality of forests and ecosystems to support the movement of species between habitats (Bierwagen 2007), and overall ecosystem adaptation to endemic or global change.

This study further supports the body of literature stating that development pressures resulting in LULCC, particularly in areas surrounding federally protected lands, are intensifying and thus threatening core habitat of native species and connectivity of forested and natural areas. The case study sites located further away from Glacier National Park, specifically, those closer to Flathead Lake, exemplify greater rates of land cover conversion and thus more intensive land use practices. For example, these areas have less restricted land use opportunities, showed far greater rates of urban development intensity, and had greater amounts of existing urban and agriculture land cover presence than areas directly adjacent to Glacier National Park. These areas, in contrast with West Glacier and Polebridge, which are directly adjacent to Glacier National Park, exemplify the connection between land use management and more invasive LULCC, at least in the CCE.

I might suggest the study site of West Glacier is, to some extent, safeguarded from some of the development pressures that Whitefish and Kalispell are subject to, because of its adjacency to Glacier National Park. Because of its adjacency, I speculate that the area of Polebridge, Montana, like West Glacier, is also likely to experience land cover change due to recreational opportunities and second home development. For reference, West Glacier is about 30 miles northeast of Kalispell, and about a 30 minute commute by vehicle; Polebridge is nearly 50 miles north of Kalispell, roughly 30 miles northwest of West Glacier, and about an hour and 40 minute commute by vehicle to Kalispell. West Glacier showed greater urban land cover increase than the Polebridge area, which is located on an unpaved forest access road. Polebridge deviates from many of the trends I found across the CCE in terms of the magnitude of LULCC, which demonstrates how spatial adjacency to other urban areas, can influence LULCC when adjacency to federally protected areas is kept constant.

These LULCC trends compare to other trends occurring in areas around the world with high amenity value (DeFries, Hansen et al. 2007), especially to areas in the U.S. Rocky Mountains (Rasker and Hansen 2000, Selkowitz, Fagre et al. 2002). In fact, LULCC in the CCE reflect those trends reported in the U.S. Rocky Mountains over a decade ago: residential and commercial land has increased in pace and extent, development is driven by service, recreation, and information business (secondary and tertiary economies), and sprawling land use conversion from agricultural to residential is common (Riebsame, Theobald et al. 1996). I suggest that the CCE exhibits very similar trends to the East River Valley in Colorado, studied by Riebsame et al. (1996), except that the CCE still remains relatively underdeveloped.

5.2 Conclusion

It is worthwhile to note that because of the dynamic character of the CCE, our results do not explain all LULCC in the region and thus the natural environment still plays a significant role in LULCC in the CCE. LULCC in the CCE is moving towards the more general trend of urban succession, one dominated by open space and urban development. While urban development is encroaching on some of the most wild and intact ecosystems and areas remaining in the CCE, nature-based tourism remains one of the largest industries in the region (McCool, Adams et al. 2006), and is a likely driver of this change. Additionally, social pressures and demographic change can further impact landscape spatial configuration by influencing land cover patterns. Thus, towns across the CCE, much like other towns across the Intermountain West (Vias and Carruthers 2005, Rasker 2006, Silberman and Rees 2010), must choose between sustaining pastoral landscapes and the community values attached to these landscapes, or support economic development that owes itself to the amenity-driven economy (Muller, Yin et al. 2008).

Land cover conversion, as seen here, suggests that land use in the CCE has the potential to damage fragile alpine environments through increased tourism and population, overwhelm social and housing services, and distort local economies (Silberman and Rees 2010). The growth of urban areas, the net increase and outward expansion, results in decreased regional ecosystem function (Wu, Ye et al. 2013). In the face of climate change, LULCC will also have effects on ecosystem composition. Many species will have a more difficult time adapting to climate induced change in addition to the threat of posed by urban development, smaller habitat, and more fragmented landscapes. Landscape connectivity is key for the long-term success of metapopulations that exist in island-like subpopulations, and use these areas to adapt to external pressures. LULCC has significant implications for landscape connectivity for plant and animals species, especially those that use valley bottoms as corridors to travel between habitat patches. For example, one study found that grizzly bears, independent of traffic volume, used habitats “within 100 meters of roads less than expected” (McLellan and Shackleton 1988). Thus, there exists a need to evaluate opportunities for conservation and connectivity across the Intermountain West. Our study can help to identify where landscape conservation efforts need to be prioritized given the expansion of developed areas around areas of high amenity value.

Future research priorities might include specific understandings of how LULCC trajectories may impact intact, native wildlands, especially in core forest areas harboring federally listed interior species. Further studies are needed to assess LULCC in this region, which can greatly aid in the protection of species diversity, ecological diversity, and the protection of the few remaining ecologically intact landscapes in North America.

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