ABSTRACT

ECOSYSTEM SERVICES IN A RURAL LANDSCAPE OF SOUTHWEST OHIO

by Meimei Lin

Grasslands provide essential services and benefits to support and maintain human populations worldwide, and regulating services may be particularly important. The first chapter of this thesis reviews the breadth of regulating services provided by grasslands, including the current measurements and known approximations. Gas regulation, climate regulation, water maintenance, soil conservation, and waste treatment are the five most important regulating services. Land use/land cover change is considered as one of the most important drivers that led to the degradation of ecosystem services. The second chapter of this thesis uses multi-temporal Landsat TM image to reconstruct land use/land cover change over time, to evaluate changes in the economic values of various ecosystem services provided by each land cover class, and to determine the causes of total economic change on ecosystem services. This study provides general useful information about the gains and losses of ecosystem services due to dynamic land use/land cover change.
ECOSYSTEM SERVICES IN A RURAL LANDSCAPE OF SOUTHWEST OHIO

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CHAPTER 1

Regulating Services and other Benefits Provided by Grasslands

Abstract

Grassland accounts for an enormous portion of the terrestrial ecosystem, which make it one of the most important biomes on Earth. Grasslands provide a variety of essential services and benefits to support and maintain flora, fauna, and human populations worldwide. In addition, there are additional services that support the functionality of grasslands including regulating services. Many of the regulating services are not incorporated into market value, but they maintain the ecological processes that are essential to human life. Therefore, a detailed analysis and description of various ecosystem services provided by grassland ecosystems will aid in the assessment of better management practices, especially the ones that are non-marketable. In this paper, I reviewed the breadth of regulating services provided by grasslands, including the current measurements and known approximations. Gas regulation, climate amelioration, water maintenance, soil conservation, and waste treatment are the top five most important regulating services provided by grasslands. Further research on the role of carbon in grassland ecosystems, and the potential for both grassland vegetation and soil under different management systems to store carbon is needed.

Keywords

Gas regulation; Carbon storage; Climate regulation; Soil retention; Water regulation; Water supply; Waste treatment;
1 Introduction

Grasslands provide a variety of essential services and benefits to support and maintain flora, fauna, and human populations worldwide (White et al. 2000), which makes it one of the world’s most useful resources to human society (Gibson 2009). Most of these benefits recognized by the public are the support of domestic livestock, thereby becoming the source of meat, milk, wool and leather products (Sala and Paruelo 1997, White et al. 2000). However, there are additional services that support the functionality of grasslands, which tend to be under-appreciated and taken for granted (Sala and Paruelo 1997, Kemp and Michalk 2007). Therefore, it is critical to assess the importance of these services and benefits provided by grasslands, especially the ones that are non-marketable.

Grassland is the ecosystem that is dominated by members of the family Poaceae. Species composition varies within grasslands but there are general consistencies in flora, plant growth form, and vegetative physiognomy (Barbour and Billings 2000). This ecosystem contains major associations including those of the steppes, velds, pampas, puszta, and prairies (Barbour and Billings 2000). There has been much confusion in the application of the term “grassland”. For the purpose of this review, the term grassland refers to any ecosystems that are dominated by grasses, including rangelands/prairies that are grazed by livestock; and semi-natural grasslands that are created and maintained by human activities, excluding savannas. Therefore, grasslands of the world include the steppes of Central Asia, the cerrado and campo of South America, the prairies of North America, and the hummock grasslands of Australia (White et al. 2000, Kemp and Michalk 2007).

Grasslands are of two origins: (i) those whose natural occurrence is dictated by climatic bounds and (ii) those which have been created through deforestation and maintained by fire or cutting or both (Shantz 1954). For example, some temperate grasslands, such as the steppes in Central Asia and the prairies in North America, occur in natural climatic bounds where the rainfall is too low to form forests. Other temperate grasslands, for example in north-western and central Europe, New Zealand, Australia, some parts of North and South America, are anthropogenic and maintained by extensive management, which is used intensively for agricultural production.
Estimations of the world’s grassland area vary due largely to diverse definitions and classification characteristics of the grassland ecosystem (White et al. 2000). Therefore, according to the aforementioned definition of grasslands, it is estimated that the world area of grasslands is $35 \times 10^6$ km$^2$, which accounts for approximately 24% of the terrestrial ecosystem (Shantz 1954, Sala and Paruelo 1997, IPCC 2000). Although they included savannas in their calculations, White et al. (2000) provided the most recent and complete estimations of the world’s grassland area. Australia, the Russian Federation, China, the United States, and Canada were the top five countries with the largest grassland area, with approximate total grassland area of $6.5 \times 10^6$ km$^2$, $6.2 \times 10^6$ km$^2$, $3.9 \times 10^6$ km$^2$, $3.3 \times 10^6$ km$^2$, $3.1 \times 10^6$ km$^2$ respectively (White et al. 2000). Because there are no savannas in these countries, these statistics are probably accurate representations of grassland area as defined here.

Although the approximate area of grasslands has decreased by less than 1% from 1700 to 1980, the spatial distribution has changed due to the conversion of grasslands to croplands and the creation of new grasslands from deforestation (Meyer and Turner II 1992). The conversion of grasslands to croplands has led to net losses of 27.4 %, 13.7%, and 26.4% for Europe, North America, and Southeast Asia respectively. Creation of new grasslands through deforestation led to increases of 10.1% and 26.2% for tropical Africa and Latin America respectively (Meyer and Turner II 1992). The Millennia Ecosystem Assessment (MEA) reported that approximately 70% of the world’s natural temperate forests and grasslands has been lost by 1950, due largely to conversion to agriculture (MA 2005). Further decline in grassland area is projected to occur between 2000 and 2050 due to human-dominated uses (MA 2005). Conversion of grasslands to agricultural land has drastically altered the condition and structure of grassland ecosystems (Kemp and Michalk 2007, Egoh et al. 2011).

In addition to the land use conversion, other impacts (e.g., global climate change, biodiversity loss, exotic species invasion, air pollution) have led to great reduction and fragmentation in natural and semi-natural grasslands (Kemp and Michalk 2007, Prochnow et al. 2009), which makes grassland the most endangered ecosystem in the world (Ceballos et al. 2010). Correspondingly, these changes to grassland ecosystems
have resulted in the degradation of many ecosystem services and benefits provided (MA 2005, Harrison et al. 2010).

The term “ecosystem services” (ES) is used to describe all the goods and services people obtain from ecosystems (Costanza et al. 1997, Daily 1997, MA 2003). ES can be divided into four categories (Figure 1): supporting, provisioning, regulating, and cultural services. While the supporting services indirectly affect people, they are the basis for maintaining the proper function of all other services (MA 2003). However, these other services do directly affect people. Provisioning services refer to the products taken from ecosystems. Regulating services are the benefits from the regulation of ecosystem processes. Cultural services are the nonmaterial gains from ecosystems. Similarly, ES of grasslands are divided into four groups (Farber et al. 2006): supporting services (e.g., nutrient cycling, pollination services, and primary production), provisioning services (e.g., domestic livestock, food crops, and genetic resources), regulating services (e.g., carbon storage, soil retention, and climate regulation), and cultural services (e.g., tourism, aesthetic and spiritual gratification).

People’s appreciation towards ES provided by grasslands is limited (Kemp and Michalk 2007). Only in the last 30 years have people started to consider the regulating services provided by grasslands, as well as the important role that grasslands play in the maintenance of multiple ES. Many of the ES, particularly regulating services, have been previously ignored because many of them have no known market value (Costanza et al. 1997, Daily 1997, Sala and Paruelo 1997).

A detailed analysis and description of various grassland ES will aid in the design of better management practices (Meyer and Turner II 1992). However, there have been few contemporary studies that look at the breadth of the services provided by grasslands (Harrison et al. 2010). Therefore, our review will be focused on regulating services that currently are non-marketable and discuss briefly the other services (supporting, provisioning, and cultural services). A review of the breadth of the services provided by grasslands, including the current measurement and known approximations, will aid in illuminating where further research is needed.
2 Regulating services provided by grasslands

This section focuses on key regulating services provided by grasslands using a selected set of service classifications by de Groot et al. (2002), including gas regulation, climate regulation, water maintenance, soil conservation, and waste treatment.

2.1 Gas regulation

Gas regulation is defined as bio-geochemical processes, influenced by both abiotic and biotic components of natural ecosystems, that maintain the chemical composition of the atmosphere (and oceans) (de Groot et al. 2002). Benefits of gas regulation include the provision of clean, breathable air, and ozone protection from diseases (e.g., skin cancer) (de Groot et al. 2002). Grasslands regulate greenhouse gases through the uptake of CO$_2$ in vegetation, the storage of carbon in soils, and the production and consumption of other greenhouse gases (e.g., CH$_4$, N$_2$O, and NO). These regulating processes contribute to global carbon cycling and climate change. Here we reviewed the literature on the estimation of carbon storage, carbon allocations between vegetation and soil pools, factors affecting carbon storage, and roles of other greenhouse gases in grasslands.

2.1.1 Estimation of carbon storage in grasslands

Grasslands store significant amounts of carbon (C), and thus have the ability to regulate CO$_2$ concentration in the air, which plays an important role in global climate change (Sala and Paruelo 1997, White et al. 2000, Gibson 2009). Although there is less carbon storage per unit area in grasslands than forests, C storage in grasslands is comparable to that of forests on a global scale due to their more extensive distribution (Gibson 2009). Grasslands store carbon in both vegetation (including both above-ground and below-ground biomass) and soils.

Global estimates of the relative amounts of carbon in vegetation suggest that grasslands probably contribute $>10\%$ of the total biosphere C store (Eswaran et al. 1993,
Nosberger et al. 2000). A few studies have estimated the amount of C that is stored in grassland vegetation on a global scale (Table 1). Large discrepancies exist due to both different data sources (e.g., including or excluding savannas) and methods (the estimates range between 27.9 Pg C and 231 Pg C). After excluding the smallest and the largest estimates, grasslands on Earth store an average of 63.3±13.5 Pg C (Mean ± SD). The estimation of carbon storage was also measured and reported regionally. The amount of C stored in grassland vegetation varied between high latitude (14-48 Pg C), mid-latitude (17-56 Pg C), and low latitude (40-126 Pg C) (White et al. 2000). Grasslands in China store 3.32 Pg C in vegetation which accounts for 4.4-11.9% of the global grassland vegetation carbon (Fan et al. 2008).

Unlike other ecosystems, grasslands allocate more carbon to below-ground biomass than above-ground biomass. In the study of grasslands in north central Alberta, Canada, Arevalo et al. (2009) found that the shoot to root biomass ratio ranges between 1.35 and 19.4 in non-grassland sites, while the shoot to root biomass ratio is 0.87 in their grassland sites. In the study of China’s 17 grasslands, Fan et al. (2008) estimated that on average the amount of carbon allocated to below-ground biomass was 22 times that of above-ground biomass. The C allocation between above-ground and below-ground vegetation varies among different types of grasslands. Fan et al. (2008) showed that the ratio of below- to above-ground carbon density among 17 Chinese grasslands ranges from 1.0 to 52.3. The below- to above- ground biomass ratio is estimated to be higher in alpine grasslands than temperate grasslands (Körner 1989, Fan et al. 2008). In addition, grassland below-ground biomass is consistently concentrated in the upper layers of the soils. In most temperate grassland ecosystems, 75-80% of the root biomass is in the top 30 cm of the soil (Jones and Donnelly 2004). According to Fan et al. (2008), 66-57% of the below-ground biomass was in the top 0-10 cm soil.

Grasslands have higher levels of carbon accrual and sequestration in soils than in vegetation, allowing them to provide a significant service toward global C storage and regulation of atmospheric CO₂. According to White et al. (2000), globally, grassland soils can store 579 Pg C, with 281, 140, and 158 Pg C in high latitude, mid latitude, and low latitude grasslands respectively (which are 20-5.8, 8-2.5, and 4-1.2 times of their
vegetation storage in the same latitude region). The carbon stored in grassland soil can account for 39% of the terrestrial carbon inventory (IPCC 2000).

Soil carbon storage includes two major components: soil organic carbon (SOC) and soil inorganic carbon (SIC). However, most studies have been focused on SOC because SOC is the main component in most ecosystems, and also because SOC is the key factor of soil fertility and vegetation production (Wang et al. 2010). In grasslands and shrub-grasslands, soil carbon in 1-3m was mainly contributed by the form of SOC (Wang et al. 2010). The terrestrial SOC contains about two times the amount of carbon stored in the atmosphere and vegetation (IPCC 2000), and grasslands account for 12% of the earth’s SOC (Schlesinger 1977). Grassland SOC accumulates from above-ground litter production, root turnover, and animal waste, and is lost through soil respiration, soil erosion, and leaching (Jones and Donnelly 2004, Cui et al. 2005). However, the carbon transfer from various sources to the soil is difficult to accurately quantify due to that input and output processes into grassland SOC occur simultaneously and at different rates depending on environmental conditions (Reeder et al. 2001).

2.1.2 Carbon sinks, carbon sources, and residence time

Grasslands regulate CO$_2$ through their major carbon pools (reservoirs with the capacity to store and release carbon) which perform as either sinks or sources. Its carbon allocation pattern determines how much of the net production is stored in different pools. The two most important pools in grasslands are soil and vegetation pools. Both can act as either a carbon sink or a carbon source depending on temporal scale, spatial scale, and other conditions. On a global scale, studies have shown that the world’s grasslands are currently an annual active sink (Thornley et al. 1991, Fisher et al. 1995), and are likely to increase greatly in the future (Scholes and Hall 1996, Scurlock and Hall 1998). Natural grassland ecosystems may contribute as much as 20% of total terrestrial production to provide an annual sink of about 0.5 Pg C (Scurlock and Hall 1998), emphasizing again their significant influence on global climate change and the carbon cycle (Hall and Scurlock 1991, Hall et al. 1995, Sala et al. 1996).
Grasslands store more carbon in soils than in vegetation. This significant character is a key factor to global carbon cycle in terms of the residence time of carbon. Most grassland SOC has intermediate residence time (10-15 yr) without disturbances, while some types of grassland may have turnover time from hundreds to thousands of years (Jenkinson 1990, Paul et al. 1997, Post and Kwon 2000, Jones and Donnelly 2004). The stability and long residence time of SOC in grasslands means that most soil carbon was stored in the soil in a stable form at large temporal scale (Jastrow et al. 1996). However, carbon stored or sequestered in the above-ground biomass is in a non-permanent form of carbon storage due to the susceptibility to disturbances (e.g., fire and grazing), senescence, and decomposition.

2.1.3 Impacts on carbon storage in grasslands

One of the most important factors that change the carbon stock in grasslands is grazing. Generally, slight or moderate grazing intensity is beneficial to maintaining production of above-ground biomass and biodiversity compared with grazing exclusion (McNaughton 1979, Hik and Jefferies 1990, Collins et al. 1998, Frank et al. 2003). However, many studies have shown that overgrazing leads to a reduction in grassland productivity (Stuth et al. 1996, Wang and Ripley 1997, Shang et al. 2003). High levels of grazing cause diminished productivity of both above- and below-ground biomass due to the direct removal of biomass by animals and the reduced potential to produce new biomass (Han et al. 2008). Grazing also affects the allocation ratio between above- and below-ground biomass, as below-ground vegetation responds differently from above-ground vegetation under the same grazing intensity (Cui et al. 2005). The root:shoot ratio of carbon allocation also responds differently to grazing levels (Caldwell et al. 1981, Holland et al. 1992, Biondini et al. 1998, Reeder and Schuman 2002). For example, in the grassland in Inner Mongolia, a slight to moderate decrease in root:shoot ratio occurred in 2005 under intensive grazing, while no significant change in root:shoot ratio was observed under less intensive grazing level in 2006 (Gao et al. 2008).
The impact of grazing on soil C storage is more complex and less consistent than the impact on biomass carbon because different grazing intensities have varied effects on soil C storage due to different soil characteristics (physical, chemical, and biological), vegetation types, grazing animal species, and other factors (Lavado et al. 1995). Because of this complexity, it is still unclear how soil C storage responds to different grazing pressures (Conant et al. 2001, Reeder and Schuman 2002). For example, significantly higher soil C has been reported in grazed pastures compared to that in non-grazed enclosures or overgrazed lands (Rice and Owensby 2001, Reeder and Schuman 2002, Liebig et al. 2005). Han et al. (2008) found that SOC (the main component of soil C storage) was higher in low grazing grasslands (35.5, 25.2 and 21.7 g kg\(^{-1}\) for 0–10, 10–20 and 20–30 cm soil, respectively) as compared to highly grazed grasslands (29.5, 14.9 and 15.0 g kg\(^{-1}\)). However, it has also been reported that grazing either has no effect (Dormaar et al. 1977, Milchunas and Lauenroth 1993, Wang and Ripley 1997) or causes a decrease in soil C storage (Frank et al. 1995, Derner et al. 1997, Snyman and Du Preez 2005). For example, Cui et al. (2005) found that SOC content and spatial heterogeneity in the Inner Mongolian steppe did not change under light grazing intensity over a 20-year period. Frank et al. (1995) showed that a grazing-induced change in community structure did not necessarily lead to decreased soil C storage. Other studies in Inner Mongolia suggested that SOC is sensitive to grazing (Xiao et al. 1996, Wang and Ripley 1997, Ojima et al. 1999). These inconsistent effects on soil C storage were due to different types of grazing practices, soils, climates, and the diversity of plant species, (Schuman et al. 2001, Derner et al. 2006).

Climate change is another factor that is likely to affect grassland C storage capacity and CO\(_2\) regulation. Both model simulations and field experiments have showed inconsistent results for the effects of climate change on C storage. Using the CENTURY model, Ojima et al. (1993) and Parton et al. (1995) predicted a substantial loss (3-4 Pg C after 50-year) of SOC for all grassland areas in response to temperature increases of 2-5°C, which was caused by higher decomposition rates. However, in contrast, Thornley & Cannell (1997) and Cao & Woodward (1998) predicted a net increase of soil C in temperate grasslands. This is a result of accelerated photosynthesis induced by higher
CO₂ that excelled the enhanced respiration prompted by higher temperature. Using a more detailed simulation model called PaSim, Riedo et al. (2000, 2001) suggested that an increase in temperature of 2°C will have variable effects on the labile SOC content of cool temperate grasslands in Switzerland. Experiments suggest that soil C pools may be depleted in the short-term but increased in the long-term with climate warming (Giardina and Ryan 2000, Thornley and Cannell 2001). Those experiments suggest that, in the short-term, higher temperature causes soil carbon decreases due to accelerated microbial respiration, but in the long-term more C is sequestered because of higher NPP and accelerated physicochemical ‘stabilization’ reactions.

2.1.4 Other greenhouse gases

Research of trace gases has largely focused on CO₂, though the importance of other greenhouse gases such as methane (CH₄), nitrous oxide (NO and N₂O), and others have also been recognized recently. Global grasslands have significant services and functions that regulate these greenhouse gases.

Grasslands can produce CH₄, NO and N₂O, which must be taken into account when the overall capacity of grasslands to regulate greenhouse gases is assessed. CH₄ is produced in grasslands mainly as a by-product of ruminant digestion of livestock due to the action of microbes (bacteria, fungi and protozoa) in the rumen. NO and N₂O is formed in the soil through nitrification and denitrification controlled by soil factors including moisture content, temperature, fertilizer additions, pH, organic matter content, and nitrate (NO₃⁻) and ammonium (NH₄⁺) (Saggar et al. 2004, Hopkins and Del Prado 2007, Schils et al. 2007). As a result, most of grasslands were often estimated to be net sources of CH₄ or NO or N₂O (Levy et al. 2007, Wang et al. 2009, Tenuta et al. 2010), though the source strength is much less than those of croplands and wetlands. The emission of CH₄, NO, and N₂O may represent a significant trade-off for the observed carbon sequestration effect when the greenhouse warming potentials of CH₄, NO, and N₂O are taken into account (Saggar et al. 2004, Ammann et al. 2009), though the magnitude of their flux is normally small (Carol Adair et al. 2009).
Grasslands also have the ability to consume CH$_4$, NO, and N$_2$O. Some grassland can even perform as a sink to these greenhouse gases. Grasslands are able to uptake CH$_4$ from the atmosphere by diffusivity of CH$_4$ into the soil and by the biological activity of methanotrophs (Dijkstra et al. 2011). Semiarid grasslands have been shown to be important sinks of CH$_4$ on a global scale, removing between 0.5 Pg and 5.6 Pg of CH$_4$ from the atmosphere each year (Mosier et al. 1991). In addition, there are also numerous studies of net negative fluxes of N$_2$O (i.e. fluxes from the atmosphere to the soil) in the literature (Blackmer and Bremner 1976, Ryden 1983, Chen et al. 1997, Yamulki et al. 1997, Glatzel and Stahr 2001), that is, the grasslands soils also have mechanics consuming N$_2$O and can act as a sink helping account for the current imbalance in estimated global budgets of N$_2$O (Chapuis-Lardy et al. 2007), though so far research has concentrated on the net N$_2$O production.

Agricultural usage and grazing change have important impacts on the regulation of CH$_4$, NO, and N$_2$O by grasslands. Many studies have shown that conversion of cropland to perennial grassland reduces emissions of CH$_4$ from upland soils (Keller et al. 1990, Dorr et al. 1993, Parashar et al. 2001), while conversion of grasslands into croplands increases emission of N$_2$O (Gelfand et al. 2011, Roth et al. 2012). Grazing can increase the emission of CH$_4$, NO and N$_2$O significantly. In New Zealand, CH$_4$ and N$_2$O contribute to 37.5% and 17.4% of the greenhouse gas emissions inventory due to intensive grazing of their pastures and relatively low levels of heavy industrial activities and vehicular CO$_2$ emissions per unit area of land (Saggar et al. 2004). N addition by fertilizer to grasslands is also reported to trigger large N$_2$O emission pulses in grasslands of Europe (Flechard et al. 2005, Ammann et al. 2009), Inner Mongolia (Peng et al. 2011), and North America (Avrahami and Bohannan 2009).

Climate change and increased CO$_2$ will have effects on the flux of these greenhouse gases as well by affecting microbial processes, but the impacts and underlying drivers are still unclear due to limited studies. In a Free-Air CO$_2$ Enrichment (FACE) experiments on permanent grassland, Abbasi et al. (2011) found that elevated CO$_2$ increased the CH$_4$ oxidation by 49% compared to ambient CO$_2$, and simulated a third source of N$_2$O. In another 4-year experiment, Cantarel et al. (2012) suggested that climate change modifies the microbial processes and induced change of N$_2$O fluxes.
2.2 Climate regulation

The complex interactions of regional and global circulation patterns with local topography, vegetation, albedo, and the configuration of water bodies together determine the local weather and climate. Research has already demonstrated that the increase in the atmospheric concentrations of greenhouse gases (e.g. CO₂, methane, and nitrous oxide) in the atmosphere can lead to the disruptions of global climate systems (Mitchell et al. 1990). As one of the critical services provided by grasslands, gas regulation (as discussed above) contributes significantly to the amelioration of global climate change.

Moreover, grasslands can regulate regional and global climate through the moderation of soil temperature and ambient temperature provided by the grassland plant species. Land transformation from grasslands to croplands, as well as various grazing intensities (e.g., light-, moderate-, and over-grazing) can alter climate at different scales (Sala and Paruelo 1997). Sala and Paruelo (1997) reported that a shift from natural grasslands into croplands can change the energy balance of a region, in turn, can alter the mesoscale climate. Field investigation discovered that light to moderately grazed grasslands can lead to increase in albedo (more sunlight is reflected) due to less plant cover and more bare soil. However, overgrazed grasslands can be easily invaded by shrub, which ultimately lead to the increase in plant cover and decrease in albedo (less sunlight is reflected) (Aguiar et al. 1996).

2.3 Water maintenance (water regulation and water supply)

Water supply and water regulation maintenance is one of the most valuable regulating services provided by grasslands (Egoh et al. 2011), which is important for agriculture, industrial and domestic water use (Kemp and Michalk 2007). Water regulation is essential in water distribution along the surface because too little or too much runoff can present serious problems (de Groot et al. 2002). Water regulation includes increased water percolation which decreases the speed of runoff and increases water purification, thereby retains the water in the system, buffers extremes in discharge.
of rivers, regulates the channel flow (de Groot et al. 2002). Water supply is maintained through the filtering, retention, and storage functions of grasslands. Performing primarily by the biological component of the grasslands, filtering results in water purification. While topography, sub-surface characteristics, hydrological cycles, and the ecosystem contribute to water retention and storage through infiltration, groundwater recharge, and runoff reduction (de Groot et al. 2002).

Grassland plant species and soil biota filter water thereby reducing the delivery of sediments, nutrients, and chemicals into the water supply (de Groot et al. 2002). Grasslands dominated by perennial species can prevent soil loss and filter water better than annual species due to deeper root systems and longer growing seasons, especially with high intensity short-duration storms (Kemp and Michalk 2007). Conservation buffers employ this filtration to reduce soil erosion and maintain water quality. In the US, since 1988, buffer strips are an approved USDA cost-share practice under the CRP program of the Food Security Act of 1985. Buffer strips are areas of permanent vegetation, including riparian buffers, filter strips, grassed waterways, contour grass strips, cross-wind trap strips, and vegetative barriers (USDA-NRCS). The USDA National Resource Conservation Service (USDA NRCS) claims buffers have the potential to remove 75% or more of sediment and 50% or more of nutrients from the water (USDA NRCS 2012). In addition, buffer strips have been shown to reduce runoff transfer of pesticides and pollutants (Patty et al. 1997). The reduction of immediate surface runoff and the storage of water in groundwater contribute to the water supply (Kotze and Morris 2001, Egoh et al. 2011). Grass buffer strips have been experimentally shown to reduce surface runoff by 43 to 99% (Patty et al. 1997, Tingle et al. 1998).

infiltration of soluble compounds into the soil profile and improve plant and soil absorption of dissolved materials.

Experimental evidence shows the decrease flow rate has contributed to the removal of at least 66% of sediment, at least 87% of suspended solids, and 22 to 100% of soluble nutrients (Patty et al. 1997, Rankins et al. 2001). In 2011 the grasslands and wetlands enrolled in the Conservation Reserve Program were credited with reducing the loss of nitrogen and phosphorous from adjacent agricultural lands by 623 million pounds and 124 million pounds respectively (USDA NRCS 2012). The NRCS claims buffers have the potential to remove 50% or more of certain pesticides. Grass buffer strips have been experimentally shown to reduce pesticides, pollutants, and their metabolites by 44 to 100% (Patty et al. 1997, Rankins et al. 2001). Additionally the presence of pathogens from livestock and poultry operations threatens water quality. NRCS claims buffers can offer a 60% or greater reduction of certain pathogens in the water supply (USDA NRCS 2012).

2.4 Soil retention/conservation

Soil retention allows for maintenance of land as well as decreased sedimentation in waterways and clouding of water. The services provided by soil retention contribute greatly to maintain agricultural productivity (de Groot et al. 2002). Many studies have shown that grassland play an important role in stabilizing and retaining soil in the system due to its dense and permanent vegetation cover, thereby protecting the soil surface from compaction and erosion (Melville and Morgan 2001, Evans and Boardman 2003, Fiener and Auerswald 2003, Fullen et al. 2006). Kemp and Michalk (2007) reported that perennial species in temperate grasslands can efficiently utilize water and prevent soil erosion and nutrient loss due to deep root systems and long growing seasons. Grassed buffer strips are commonly employed to combat soil erosion and nutrient loss with varied effectiveness (Kohnke and Bertrand 1959, Dillaha et al. 1989, Rose et al. 2003, Toy et al. 2005, Fullen et al. 2006).

2.5 Waste treatment
Through dilution, assimilation and chemical re-composition, natural ecosystems, to a limited extent, have the capacity of storing, recycling and reducing certain amounts of pollutants and human waste (de Groot et al. 2002). Grasslands, as highly dynamic ecosystems, allow for the reduction of waste and nutrients further maintaining soil fertility and decreasing eutrophication. Hillman et al. (2003) found that grassland can process material that arable land cannot, which makes it a critical strategic repository for sewage sludge. Moreover, there is no time restriction in using grassland to receive sewage sludge, while arable land can only be used seasonally after crop harvest. It was estimated that approximately 439,705 tonnes dry matter is applied annually to agricultural land in the UK, with 37% being applied to grassland (Carrington et al. 1998). Furthermore, it’s easier to dump sewage sludge in grassland than cultivated land. With grassland, the sewage sludge can be applied directly to the surface while in cultivated land biosolids need to be injected into the soil (Carrington et al. 1998).

Many researchers have demonstrated the effectiveness of grassed buffer strips in removing pesticides, nitrate and soluble phosphorus (P) compounds from runoff water (Doyle et al. 1977, Karr and Schlosser 1978, Walter et al. 1983, Barker and Young 1984, Schwer and Clausen 1989, Osborne and Kovacic 1993, Correll 1997, Patty et al. 1997, Lee et al. 2003, Borin et al. 2005, Lovell and Sullivan 2006). Doyle et al. (1977) compared the effectiveness of grass strips with that of forest strips and found that both types of filter successfully reduced levels of bacterial indicators of fecal contamination. Schwer and Clausen (1989) evaluated a grass filter for treating dairy milkhouse wastewater and discovered that the concentrations of total solids, total P and Total Kjeldahl Nitrogen were reduced by 92, 86 and 83% in surface run-off, and 97, 92 and 93% in subsurface run-off. Barker and Young (1984) and Walter et al. (1983) carried out similar studies and obtained similar results. Patty et al. (1997) investigated the value of grassed buffer strips and found out that grassy buffers are effective in storing and recycling pollutants and nutrients. Depending on the width of grassed buffer strips (6, 12 and 18 m), runoff volume was reduced by 43-99.9%, suspended solids by 87-100%, lindane losses by 72-100%, atrazine and its metabolites by 44-100%, nitrate losses by 47-100% and P losses by 22-89% (Patty et al. 1997). Borin et al. (2005) also found that
grassy buffers can trap sediments and P bound components therefore improving pollution control.
3 Other services and benefits provided by grasslands

3.1 Supporting services provided by grasslands

Grasslands also have supporting services, which are needed to maintain the other three services. They differ from provisioning, regulating, and cultural services in that their impacts on people are often indirect or occur over a very long time (MA 2005). Supporting services of grasslands include nutrient cycling, primary productivity, pollination, habitat, and hydrological cycles (Farber et al. 2006). Nutrient cycling involves the processing and storage of nitrogen and phosphorus. Primary productivity allows for the conversion of energy to form available to other tropic levels. Grasslands are habitat for a variety of insects that can provide pollination services to surrounding crop lands. Grasslands are important habitat for wildlife including many bird species. For example, they can provide nesting, brood, and winter cover to grassland-dependent bird species.

Biodiversity, especially functional diversity, is important of stability of the ecosystem services available. Biodiversity can be represented in the flora; can serve as buffer in times of fluctuation, disturbance, plant pathogen infestations, and climatic change; can increase provisioning services (through increases in productivity); and can increase regulating services. Biodiversity of insect populations can provide continued availability of natural pollinators (which is of increased importance with the rising problem of colony collapse in honeybees). Grasslands can support biodiversity of animals (which can have many impacts including the reduction of zoonose which translates to the reduction of disease risk). In addition biodiversity is generally found to be aesthetically pleasing.

The distribution of grasslands impacts the ecosystem services provided and the scale and distribution of these services. Land use change impacts the ecosystem services available. Further research is needed to assess the changes in ecosystem services association with land use changes. Trade-offs between services will occur with land use change. Management and decisions of land use change need to provide for the greatest service flow from an ecosystem.
3.2 Provisioning services provided by grasslands

Provisioning services refer to the products obtained directly from ecosystems, including milk, wool, fresh water, medicine and genetic resources. Provisioning services provided by grasslands are highly variable depending on location and weather condition. Most of the grassland ecosystems are used as grazed rangelands to support the production of domestic livestock. It was estimated that the world’s grasslands can provide forage to support over 1800 M grazing livestock units and wildlife populations (World Resources Institute 2000). The large numbers of domestic livestock supported by grasslands are the primary source of meat, milk, wool, and leather products for humans (White et al. 2000).

Moreover, grasslands provide habitats for wild herbivores. Grasslands are used as breeding, migratory, and wintering habitat by wild herbivores, which normally share the rangelands with livestock animals. Knapp et al. (1999) reported that the American bison plays a keystone role in the development and maintenance of the US tallgrass prairie. The major cereal crops on earth, including wheat, corn, maize, rice, millet, and sorghum, which are critical for human being to sustain life worldwide, originate from grasslands (White et al. 2000, Gibson 2009). Many grasslands are important genetic resources, which are potentially useful in the improvement of our crops and the increase of pharmaceutical products (White et al. 2000). Williams et al. (2000) reported that about 30% of all plants sold in South Africa’s traditional medicine markets grow in grasslands.

With increasing pressure to attain alternate fuel, grasslands also offer the possibility of use as biofuel. In Europe and North America, a large number of perennial grasses are used to produce biofuel as they can grow with the minimum inputs and yield high-carbon biomass (Gibson 2009). It has been stated that grassland is suitable in numerous ways as alternate fuel source (Walsh et al. 2003, Gibson 2009, Prochnow et al. 2009). Tilman et al. (2006) assessed the effectiveness of low-input high-diversity (LIHD) mixtures of native grasslands in producing biofuel. Their results showed that the bioenergy yields produced by LIHDs are 238% higher than that of monocultures. Moreover, LIHDs sequester more CO$_2$ in the soils and roots ($4.4$ Mg CO$_2$ ha$^{-1}$ yr$^{-1}$) than
CO₂ release during the conventional biofuel production (0.32 Mg CO₂ ha⁻¹ yr⁻¹), which makes LIHD a carbon negative system (Tilman et al. 2006). Therefore, grassland has the potential to be a valuable bioenergy source.

3.3 Cultural services provided by grasslands

Grasslands are important in having cultural values through recreational opportunities, aesthetic enjoyment, or spiritual benefits (Harrison et al. 2010). Recreational opportunities of grasslands include sports, bird-watching, ecotourism, and hunting. Grasslands have been used increasingly by tourists as a destination for recreational activities, such as hiking, fishing, viewing of game animals, and safari hunting (White et al. 2000, Gibson 2009). Large mammalian herbivores, grassland birds, diverse plant life, and vast open-air landscape attract people from all over the world.

In addition, grasslands can serve as educational places or to be used in scientific research as well. Throughout the prairies of the US, the preservation of religious, ceremonial, and historical sites was reported (White et al. 2000). Increasingly, grasslands are considered as culturally and spiritually important sites.

People’s appreciation towards grassland is often thought to be related to the appreciation towards species diversity. Lindemann-Matthies et al. (2010) found that the most aesthetically pleasing grasslands are those that were most diverse. Species-rich grasslands are an important focus of conservation efforts in many regions. However, with agricultural intensification, and anthropogenic disturbance, the aesthetic appreciation of grassland by people is reduced (Lindemann-Matthies et al. 2010). White et al. (2000) investigated the tourist data in countries with extensive grassland and found out that grasslands contribute greatly to economic revenues through recreation and tourism, especially safari and hunting. However, they concluded that the excessive human use and wildlife poaching can potentially lead to the degradation of grassland ecosystems and decreased capacity to maintain tourism services.
4 Conclusions

Grassland is one of the most important biomes on Earth. The ecosystem services and benefits provided by grasslands are essential to sustain all organisms and human generations. Most of our food and living resources originate from grasslands. Grasslands also provide fundamental services that support the multi-functionality of grasslands, most of which do not have known market value. Gas regulation, climate amelioration, water maintenance, soil conservation, waste treatment are the top five most important regulating services provided by grasslands, which tend to be ignored and taken for granted by the public and policy-makers.

Grasslands regulate atmospheric gas composition through carbon sequestration, methane absorption, and nitrous oxide reduction. The grasslands have a great ability in regulating greenhouse gases. They capture CO$_2$ through photosynthesis and store them in both biomass and soil, with soil as the biggest and most stable carbon pool. They also have emission of CH$_4$, N$_2$O, NO, but they also have mechanics consuming N$_2$O, NO, and CH$_4$, and soma arid, semi-arid grasslands also act as a net sink of CH$_4$. Account of greenhouse effect of all greenhouse gases, global grasslands are key greenhouse sinks, and very important in terms of climate change mitigation. Proper management (especially grazing policy) of grasslands is needed to reduce the release CH$_4$, N$_2$O, NO, and protect loss of soil carbon and degradation of grassland vegetation.

Climate regulation provided by grasslands at all scales is either through greenhouse gas regulation or through the moderation of soil temperature and ambient temperature provided by grassland species. Grasslands also maintain the water quality and distribution, as well as control soil erosion and nutrient loss from water runoff. Moreover, grasslands can store and recycle pollutants from waste.

Therefore, simply maximizing the production of grasslands as in the past is not going to resolve the problems we are facing today. In order to gain a sustainable development, ES provided by grasslands should be emphasized. Great efforts need to be made to attract public attention to improve their recognitions of services and benefits supplied by grasslands, upon which all living organisms rely (Kemp and Michalk 2007).
CHAPTER 2

Using Remote Sensing and GIS to Analyze Land Use/Land Cover Change for Its Impacts on Ecosystem Services

Abstract

Humans have dramatically changed the structure and function of ecosystems, as well as the benefits people can extract from ecosystems. Local land use/land cover change has been recognized as one of the most important drivers. However, few local-scale studies have evaluated the consequences of land use/land cover change for its impacts on ecosystem services. In this study, we used multi-temporal Landsat Thematic Mapper (TM) image to reconstruct land use/land cover changes during 1987 and 2010 in Butler County, an agricultural county in southwest Ohio, USA. In addition, we used previously published ecosystem service value coefficients to evaluate changes in the economic values of various ecosystem services provided by each land use/land cover class. Finally, we determined the causes of total economic changes on ecosystem services. Principal Component Analysis (PCA) based change detection indicated that the major land use/land cover change occurred in forested area to agriculture. The total ecosystem service value of Butler County was approximately US$ 41.8 million per year in 1987 and US$ 41.0 million per year in 2010. There was a net loss of US$ 0.8 million per year. Over a 23-year period, the cumulative loss of ecosystem service value reached US$ 9.2 million. The decline in ecosystem service value was mainly caused by the loss of forest (US$ 3.9 million). The increase of ecosystem service value caused by increases in agriculture, grassland, and open water was too small to offset the decrease. Our study provides general useful information about the gains and losses of ecosystem service due to dynamic land use/land cover changes.

Keywords

Ecosystem services (ES); Change detection; Unsupervised classification; Principal Component Analysis
1 Introduction

For millennia, natural ecosystems have provided human societies with a range of goods and benefits that support and enhance human well-being (Daily 1997). However, a rapidly increasing human population, human-driven modification of Earth, and increased per capita global consumption have changed ecosystems more rapidly and extensively than ever before, causing a series of environmental problems (Vitousek 1997, MA 2005, Turner et al. 2007, Seppelt et al. 2011). It not only affects the ecological properties of ecosystems, but also impacts the supply of ecosystem services (ES) that people can take from nature (Kremen and Ostfeld 2005, Diaz et al. 2007).

There are several interacting driving factors that have directly or indirectly led to the changes in ecosystems and the services they provide, as well as changes in human well-being (MA 2005). Land use/land cover change has long been recognized and highlighted as one of the most important factors that directly alter landscape patterns and the terrestrial ecosystems (Vitousek 1997, MA 2005, Reyers et al. 2009). Changes in land use/land cover (LULC) are long term alterations of the habitat and can alter the biogeochemical cycles, hydrology, and soil properties of an ecosystem (Reyers et al. 2009). Human influences are now so great that over 75% of the earth’s land surface has been transformed (Ellis and Ramankutty 2008). Since the establishment of agriculture, natural habitats have been converted into row crop cultivation, pastureland, and urban settlement (Grau et al. 2003). A potential consequence of this type of land conversion is likely to be the abandonment of agricultural lands, leading to forest recovery (Grau et al. 2003), as well as exurban growth (Brown et al. 2005).

A lot of attention has been paid to the study of the consequences on ecosystems and ES caused by LULC change. Most of the research is looking into the consequences on a global scale, while not much has been done on a local-scale to evaluate the effect of LULC change in terms of the delivery of ES (Hu et al. 2007, Reyers et al. 2009). Therefore, it is necessary to investigate information on local-scale consequences of LULC change over multiple ES.
To obtain LULC change information, change detection has proven to be quite useful and practical. There are many change detection techniques: PCA, image differencing, and post-classification are the most popular and accurate methods (Lu et al. 2004, Deng et al. 2008). Successful data collection on current and historical satellite images is another key factor in the application of change detection. Remotely sensed data are considered and recommended by many researchers as one of the most common data sources in the application of change detection (Singh 1989, Jensen 1996, Yang and Lo 2002, Deng et al. 2008, Abd El-Kawy et al. 2011). Once images have been obtained, an image classification technique needs to be chosen.

Both supervised and unsupervised image classification methods have their own advantages and disadvantages, and have been used widely in various studies of LULC classification (Jensen 1996). Supervised classification requires a priori knowledge of the study area (user creates signatures). However, unsupervised classification method using the Iterative Self-Organizing Data Analysis Technique (ISODATA) algorithm is more suitable for this study because it doesn’t require extensive knowledge of the study site (signatures are generated by users). Of all the clustering parameters when using ISODATA unsupervised classification, the number of classes are the most crucial, as it affects the capability of an ISODATA classifier for capturing most of the LULC variability from the image being analyzed. Typically, a larger number of clusters, say 50 or more, is used to ensure adequate data representation.

Despite increasing appreciation of ecosystem functions and services, they are often taken for granted and under-appreciated in policy-making. Most of ES do not have a clear market value. Therefore, evaluating them into economic value has proven to be difficult (Daily 1997, Balmford 2002). Since the release of the Millennia of Ecosystem Assessment (MEA), the economic valuation of ES has become a popular topic and active research area for understanding the multiple benefits provided by ecosystems. Peter et al. (1989) calculated the total financial value of an Amazonian rainforest and provided suggestions for conservation and sustainable exploitation of Amazonian forest. Most notably, based primarily on previously published studies, Costanza et al. (1997) estimated the economic value of 17 ES for 16 dominant biomes. Thereafter, their data on ES value
for each biome has been used as the first approximation when attempting to translate ES into economic values (Kreuter et al. 2001, Zhao et al. 2004, Hu et al. 2007, Li et al. 2010).

The goals of this study are to: (i) test the hypothesis that the economic value of ES provided by Butler County has changed between 1987 and 2010, (ii) quantify the magnitude and direction of that change, and (iii) determine the particular shifts in LULC that most strongly contribute to that change. We used Geographical Information System (GIS) and Remote Sensing techniques to reconstruct local LULC from 1987 to 2010, estimated LULC changes by using PCA-based change detection, and quantified the economic value of ES changes across LULC changes in terms of US dollars.
2 Methods and Materials

2.1 Study area

My study area is Butler County, located in southwestern Ohio, USA (Figure 2). It is home to more than 368,130 people with a population density of 304/ km² (US Census Bureau 2010). It occupies a total area of 1,217.3 km². The annual maximum average temperature is 17°C, while the annual minimum average temperature is 6°C. Based on a 30-year record (1961-1990), the average annual precipitation is approximately 990 mm, and the monthly average precipitation is 83.82 mm (Bartels et al. 1993).

The upland surface of Butler County is a broad, flat, and gently rolling plateau, which makes it an ideal region for farming (Klaer and Thompson 1948). With the exception of the cities of Hamilton and Middletown, the whole Butler County is largely agricultural. In 1940, approximately 90% of the total area in Butler County was farm lands (US Census Bureau 1942). Livestock, corn, soybeans, and hay were the principal sources of farm income (Klaer and Thompson 1948). In SW Ohio, 20-60% of rural lands have been converted to exurban development from the Ohio Valley Metropolitan region (Brown et al. 2005, Grimm et al. 2008). There are several large valleys in Butler County, including the Ohio, Miami, Little Miami Rivers, and Mill Creek (Klaer and Thompson 1948). The Miami River is the principal stream in the region.

Several USDA conservation programs were established by the federal government, which include land retirement programs, working land programs, and agricultural land preservation (US Department of Agriculture 2009). The Conservation Reserve Program (CRP) is the largest conservation program, which was established through the 1985 Farm Bill. CRP was designed to set aside highly erodible and environmentally sensitive acres of cropland to grassland under 10-15 year contracts. As compensation, landowners receive an annual rental payment for shifting marginal lands from production into semi-natural grasslands and buffers (US Department of Agriculture 2009).
The impacts of cultural and socio-economic differences on LULC lead to a mosaic landscape of agricultural land, urban settlement, and recovering forest and grassland. These landscapes vary in their market values, level of biodiversity, and ES.

2.2 Data collection and preprocessing

To eliminate the external impacts (such as different sun angles, atmospheric conditions, seasonal and phenological properties), the ideal image needs to have the following characteristics: produced from the same sensor, same spatial and radiometric resolution, acquired on the anniversary or near the anniversary date, and the same location. The satellite data used in this study are two full-scene cloud-free Landsat-5 Thematic Mapper (TM) (Table 2) images acquired on 24 September 1987 and 23 September 2010, respectively. Both images have been geometrically referenced to World Geodetic System (WGS) 84 datum and spheroid projection at a spatial resolution of 30 meters. A relative radiometric normalization was applied to both images to eliminate the radiometric discrepancies. Both images were converted to reflectance using the COST model, which reduces image haze effects (Chavez 1996). Using GIS tools, these two preprocessed images were extracted by mask with the boundary of Butler County to confine the Landsat TM image to the study area.

2.3 LULC change information enhancement using multi-temporal Principle Component Analysis-based approach

After the geo-rectification and radiometric normalization, the preprocessed images were enhanced by Principal Component Analysis (PCA), which transforms a set of complex correlated variables into a new set of uncorrelated variables (the principal components). A PCA-based change detection has been proven to be accurate and efficient in maintaining and capturing the maximum LULC change information (Deng et al. 2008). There are two general ways of using PCA analysis in remote sensing. The first one is to perform PCA analysis separately on each image to produce a new image. Then the selected band from one image can be compared with the same band from the other
image such as image ratioing, image differencing, vegetation index differencing. In our study, the second method was used. The two images of Butler County were stacked using the ERDAS Imagine 9.3 software to produce a composite multi-temporal image with a total of 12 bands. PCA was applied to this combined dataset of the original multi-spectral image to produce the principal components (PCs). Twelve new PCs were ordered so that the first few retained most of the variations that were present in both images. After careful examination, most of the change information was demonstrated in the first four components (Figure 3).

2.4 LULC change information extraction and label using unsupervised classification

The enhanced LULC change image was produced and ready to be extracted using image classification. Based on the knowledge of the study site, unsupervised classification was chosen. I used the same land cover system that was used in Costanza et al. (1997). Hence, prior to image classification, it was decided that LULC classes of the study area should be referred to as built-up, agriculture, open water, forest, and grassland.

Unsupervised classification methods was first performed on the PCA-enhanced multi-temporal change image to produce an unlabeled cluster map and provide a basic set of spectral classes for further analysis. To find a suitable number of clusters for my study, I tried 40, 50, 60, and 70 clusters to see which number has the best suitability in differentiating between classes. Sixty clusters were enough to distinguish each LULC class. By combining aerial photos, field GPS points, and other ancillary data, all 60 clusters were assigned into two major classes, change or no-change. Then, I recoded the produced image. No-change classes were recoded as 0, and change classes were recoded as 1. The recoded image was combined with the PCA-enhanced multi-temporal image to mask out no change classes and produce the image with only change classes. The result image from the mask out was run through another unsupervised classification to produce another unlabeled cluster map. Forty clusters were chosen this time. Finally, the
ancillary data were combined to decompose and label the image into different classes based on their changes from 1987 to 2010 (“from-to”).

2.5 Accuracy assessment

Finally, the overall accuracy of classification was calculated to justify the application and distribution of the results. Before any further application of the classified LULC change map, it is critical to know the validity and accuracy of the results. After the post-classification refinement, the classified LULC change map for Butler County from 1987 to 2010 was assessed using accuracy assessment in ERDAS IMAGINE 9.3 software. It has been well established that an error matrix and a kappa analysis are useful and common tools for accuracy analysis, and they have been considered the standard reporting procedures by many researchers (Congalton 1991). Therefore, an error matrix and a kappa analysis were used to assess the overall accuracy of classified LULC change map.

An error matrix is a square table with rows and columns set out to represent the referenced data and classified data assigned to each LULC class (Table 3). The rows usually record the reference data while the columns show the classified data from the generated image. From the error matrix, one can easily tell how many pixels are correctly classified into its corresponding class, and how many are not. The overall accuracy was calculated by dividing the total number of correct pixels (in this case, the sum of the major diagonal) by the total number of error matrix pixels. In addition, for each individual LULC class, the “producer’s accuracy” and the “user’s accuracy” were generated. The producer’s accuracy is computed by dividing the total number of correct pixels in one LULC class by the total number of pixels of that class as derived from the reference data (Congalton 1991). Dividing the total number of correct pixels in a category by the total number of pixels that are classified in that category gives the user’s accuracy (Congalton 1991).

Kappa analysis is another measure of accuracy. Its usage was first proposed in Congalton et al. (1983) and was then recommended by many researchers when
performing accuracy assessment in remotely sensed data (Congalton 1991). To get a sound result from these analysis techniques, several factors need to be considered. One of the most important steps in accuracy assessment is the unbiased collection of ground-truth data. There are two major sampling strategies: to create/add random points; to import user-defined points. I combined both sampling strategies to provide optimum balance between statistical validity and practicality.

Another critical factor in accuracy assessment is the sample size (i.e., the total number of referenced points, including both random and user-defined points). Generally, a minimum number of 50 points per LULC class is enough in an error matrix (Congalton 1991). However, the total number of sample points can be adjusted based on the relative importance of that LULC class. For example, if a specific LULC class is the objective of a study, the minimum number of its samples can be increased to 75-100, or even more. A total of 887 sample points were collected to assess the accuracy of this study.

2.6 Assignment of ES values

In order to analyze the impacts of the LULC change on the delivery of ES in terms of ES values, the economic values of each different LULC change classes should be assigned based on Costanza et al (1997). ES values were adjusted into 2012 US$ to account for inflation, using individual annual inflation rates for each year.

The economic value of the ES derived from each LULC class was calculated by multiplying the estimated area of each LULC class by the value coefficient of the appropriate biome (Table 4). The total economic value of all ES in Butler County was the sum of the values of all LULC classes.
3 Results

3.1 Overall accuracy and Kappa coefficient

The overall accuracy of the LULC change detection was 87.95% and the Kappa coefficient was 0.87 (Table 3). The result is consistent with the USGS classification criteria of 85% or higher (Anderson et al. 1976). Assessment of the producer’s accuracy (Table 3) revealed that 10 out of 17 classes had over 85% accuracy, including three unchanged classes and seven changed classes. The percentage of accuracy for change from forest to grassland/shrub and change from forest to water is 100%. Four classes showed at least 80% accuracy and two classes showed 76% accuracy. For classes with accuracy of at least 80%, the major spectral confusion was between grassland/shrub and agriculture, built-up and agriculture. The spectral characteristic of grassland/shrub is somewhat similar to that of agricultural land. Change from grassland/shrub to built-up showed an accuracy of only 12.50%. The reasons are twofold: this class only represents a very small area (63.27 ha) and four sample points were incorrectly classified. Most of the confusion happened between change from grassland/shrub to agriculture and change from grassland/shrub to built-up, which is unavoidable due to the great spectral similarities between agriculture and built-up. For example, some of the parking lots in our study area were constructed from soil, which is the same material of agricultural land. These two classes were spectrally confused and cannot be easily separated by just using unsupervised classification. As for the user’s accuracy (Table 3), the results were similar to those of the producer’s accuracy to some degree. Conditional Kappa statistics, which takes into account the agreement occurring by chance, showed an accuracy of over 0.80.

3.2 Nature and location of changes in LULC

The largest change in area occurred in forest, which decreased by approximately 8,966 ha from 45,011 ha in 1987 to 36,045 ha in 2010 (Figure 5, Table 5). Among all five LULC classes (agriculture, forest, built-up, grassland/shrub and open water), four classes showed a continuous increase but only one class - forest experienced a trend of
decrease. The second most affected LULC class was the built-up area, which increased by 3,477 ha from 11,370 ha in 1987 to 14,847 ha in 2010. Agriculture increased in area by 2,623 ha from 40,664 ha to 43,288 ha. Grassland/shrub increased in area by 2,356 ha from 24,186 ha to 26,542 ha, and open water increased by 510 ha from 730 ha in 1987 to 1,240 ha in 2010.

In 1987, forest was estimated to account for 37% of the study area, highest among all five LULC classes, followed by agriculture (33%), grassland/shrub (20%), built-up (9%), and open water (0.6%) (Table 5). However, in 2010, agriculture became the dominant LULC class and accounted for 35% of the study area. The percentage of forest decreased to 30% as a result of conversion to other LULC classes. Grassland/shrub, built-up, and open water increased from 20%, 9%, and 0.6% to 22%, 12%, and 1%, respectively (Table 5). Regarding the percentages of the changed area when compared with 1987, these LULC changes were emphasized (Figure 6, Table 5). The highest percentage increase occurred in open water, which showed an increase of 70%. The second highest percentage increase was the built-up area, showing an increase of 31%. Forested area decreased by 20%. Grassland/shrub and agriculture increased by 10% and 6%, respectively. These changes indicated a dramatic increase in open water and built-up area, and small increase was seen in grassland/shrub and agriculture. However, forested area still experienced a decrease due to land conversion.

Examination of land conversion among specific LULC class (Table 6) is beneficial to ascertain the reasons behind the observed LULC change. Table 6 showed fourteen major LULC changes occurred during 1987 and 2010, while Figure 4 showed the location of these changes. The major LULC change was forested area being converted into agricultural land by 38% (8,716 ha). The second biggest change observed was from agriculture to forest, with about 16% of all changes. The next biggest change observed was from agriculture to grassland/shrub, accounting for 14 percent of the total changes.

Changes from forest and agriculture to open water led to a total of 542 ha area increase in open water (Table 6). Even though the area increase in open water only
accounts for 2.35% of the total change, when compared with that of 1987, the size of open water increased by 70%. That is a significant increase over 23-year period of the study. There were two types of open water, one is gravel pit and the other is rivers/lakes. The size increase of open water that was used for gravel pit was 353 ha while the size increase of open water that was rivers/lakes was 189 ha.

3.3 Estimation of changes in ES values

The total ES value of Butler County was about US$ 41.8 million in 1987 and US$ 41.0 million in 2010 (Figure 7, Table 7). There was a net loss of US$ 0.8 million per year in ES between 1987 and 2010 (Table 7). Assuming a linear decline in ES, this represents a cumulative loss of US$ 9.2 million in a 23-year period. From 1987 to 2010, the decline in ES value was mainly caused by the loss of forest, which led to a direct economic loss of US$ 3.9 million per year. The increase of ES value caused by increases in agriculture, grassland, and open water was too small to offset the decrease.

Because of the highest coefficient ES value (US$ 12,193 ha\(^{-1}\) yr\(^{-1}\)), open water contributed almost US$ 2 million per year to the total ES value. As you can see from Table 6, open water was categorized into two different types, gravel pit and rivers/lakes. The reason for this separation is that if the open water is used as gravel pit, it will be considered as a human-made structure. Therefore, it has no ES value. However, if the open water is natural ecosystem (rivers/lakes), it generates enormous ES value as US$ 12,193 per hectare per year. Given these differences, it is critical to specify the open water as either gravel pit or rivers/lakes. The total ES value produced by open water was estimated to be US$ 1.9 million per year due to LULC changes (about 21% of the total ES value).

Even though the ES value produced by forest is only US$ 433 per year, because of its rapid and large area decrease, the net economic loss caused by forest decline was about 20% of the total economic loss. Since built-up area was not considered as natural ecosystem, the ES value of built-up area was estimated to be zero. Approximately 2,646
ha (about 11% of that of 1987) forested area was converted to built-up area, which led to a direct economic loss of US$ 1.15 million per year since 1987. 778 ha (about 3% of that of 1987) agriculture was transformed into urban settlement, which caused a direct economic loss of US$ 0.1 million per year. In addition, most of these land conversion to built-up area occurred in the cities of Hamilton and Middletown, where most of the industrial establishments located (Klaer and Thompson 1948).
4 Discussion

The LULC in the study area experienced dynamic changes over the 23-year period of the study (1987-2010). From 1987 to 2010, 8,716 ha of forest were converted into agricultural land while 3,739 ha of agriculture shifted into forest, resulting in a net loss of 4,977 ha in forest category. The increase in agricultural land was at the expense of ecologically important temperate forests. As discussed in the previous literature, most of the Butler County area is mainly agricultural (Klaer and Thompson 1948). The principle crops in this region are corn, soybean and hay, which contributing to local economy. We think that land conversion from cropland to forest is mainly due to the abandonment and forest recovery in Butler County. Since the establishment of agriculture, most of the natural area was converted into cropland. However, with agricultural intensification, most of the converted cropland lost its fertility and was abandoned. All this has led to the slow recovery of forest (Grau et al. 2003).

Another significant LULC change occurred in built-up category. The area increase of urbanized area in Butler County was mainly due to land conversion from forest. Rapid human population growth and higher human demands have led to urbanization and rural sprawl (Brown et al. 2005). More rural land was devoted into urban uses. Excessive loss of natural habitat to urbanization could lessen the provisioning of ES provided by these natural ecosystems, such as climate change regulation, disease prevention, water supply, as well as esthetic enjoyment. However, due to increasing human population, it is unlikely that land conversion to urban land will cease or decrease in the near future. It is highly recommended that natural ecosystems that can provide a greater level of ES should be established where there is negative land conversion.

The estimated increase of 2,356 ha in grassland represents a substantial increase in ES. Grasslands provide many ES, which makes them one of the most useful resources to human society (Gibson 2009). This increase in grassland is mostly due to conversion from agricultural land, and this was most likely due to the implementation of the USDA Conservation Reserve Program (CRP). The CRP program was initially established in
1985. Under the CRP program, farmers had the option to convert cropland into semi-natural grassland. Since LULC changes were observed between 1987 and 2010, it makes sense that we saw an area increase in grassland/shrub over 23-year period. We compared 15 known locations of CRP lands with the classified grassland in our study area. All 15 known CRP lands were in grassland category in 2010. Although more CRP lands have to be checked with grassland/shrub category to see if they were under CRP program, we think that CRP program did contribute to the area increase in grassland.

With the establishment of the CRP, land conversion from cropland to semi-natural grassland has not only purified water by reducing soil erosion, but also has increased the wildlife diversity, particularly grassland-dependent birds (Johnson and Schwartz 1993, Johnson and Igl 1995, King and Savidge 1995). Study of Johnson and Igl (1995) in North Dakota showed that there is an obvious reversion in the number of grassland bird species after the establishment of CRP land. Johnson and Igl (1995) also pointed out that CRP land plays a major role in recovering bird species diversity by providing habitat and food sources for breeding birds. King and Savidge (1995) found higher pheasant abundance in the habitats that are surrounded by higher CRP enrollment, which suggests that the establishment of CRP land also benefits the nearby cropland.

The method we used in this study to estimate ES value was based on Costanza et al. (1997) by multiplying the estimated size of each LULC class by the ES value coefficient of the biome used as the proxy for that category. However, the ES value for each biome provided by Costanza et al. (1997) has been criticized by economists for both empirical and theoretical reasons (Pimm 1997, Pearce 1998, Toman 1998, Kreuter et al. 2001). Pearce (1998) pointed out that the method used by Costanza et al. to estimate the ES value focused only on the benefits of protecting environment, not the cost of providing those benefits. Turner et al. (2003) demonstrated that Costanza et al.’s method of aggregating (at the global scale) is problematic for its uncertainty and because of the likely nonlinear ecosystem responses to changes in land cover. Even Costanza et al. (1997) presented errors and limitations of their economic valuation methods. The ES value coefficient for each land cover class in Costanza et al. (1997) paper is a global average value, which doesn’t represent real economic value in local and regional
ecosystems. Therefore, extrapolating the average value to give local values of services seems to be inaccurate. Nonetheless, Costanza et al. (1997) provided the only set of complete calculation of ES value, which can serve as the first approximation of ES value available to us.

In order to make the ES value estimation more meaningful and representative to us, it’s important that we calculate ES value coefficient using local data. However, as discussed before, most ES do not have a clear market value. It is easier to quantify the services that have direct consumptive values. Therefore, in our study, we might start by calculating some selective ES using local data. For those products that have direct consumptive value, the ES value coefficient of a particular ecosystem can be computed by utilizing the market value of the products that are produced by that particular ecosystem (Daily 1997). For example, when calculating the food supply service provided by agricultural land, one can calculate the commercial market value of the products that are harvested and sold commercially. In our study area, corn and soybean are the two major crops cultivated in farm lands. The information on annual average market price and yield per hectare of corn and soybean can be obtained from local Farmer Service Agency. Finally, the ES value coefficient of agriculture can be calculated. In this way, we can get the first approximation of ES value provided by agricultural land in food supply service.

For those services that do not have a clear market price, survey approach is one of the potential evaluation methods that can be used. Even though this method was criticized for its uncertainty, it may be the only way of measuring this type of services (Daily 1997). The information obtained from surveys might not be perfect, but it is better to gain some information than nothing at all. For example, when quantifying the ES value coefficient for grassland in providing cultural services, a survey can be applied to gather information from people who are enjoying the beauty of nature. To implement this, good survey questions need to be designed in order to obtain location specific information. People who take this survey will be asked to provide opinions and amount of money that they are willing to pay in order to preserve natural grassland.
To assess and monetize the ES value coefficient is somewhat difficult, it is critical that we develop and improve valuation methods oriented toward those ES whose values are not expressed in market prices. Despite the uncertainty and problems with ES valuation, our study generated useful information about the consequences on the delivery of ES under dynamic LULC change, which can be utilized to call attention of public and policy-maker.
References


USDA NRCS. 2012. USDA Announces Results for Conservation Reserve Program General Sign-Up. Washington, DC.


Table 1. Estimation of the amount of carbon stored in grassland vegetation on a global scale

<table>
<thead>
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<th>Estimated total area (million ha)</th>
<th>Estimated total C storage (Pg)</th>
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*including savannas
Table 2. Introduction of the sensor used in this study

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<th>Spectral coverage (µm)</th>
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Table 3. Error matrices and total classification accuracy for the classified land use/land cover change image (1987-2010)

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<td>25</td>
<td>47</td>
<td>53</td>
<td>114</td>
<td>21</td>
<td>55</td>
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<td>17</td>
<td>887</td>
<td>87.95</td>
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Table 4. Average global economic value of annual ecosystem services for five land use/land cover classes (Costanza et al. 1997)

<table>
<thead>
<tr>
<th>LULC Classes</th>
<th>Equivalent biome</th>
<th>ES coefficient (1997 US$ha(^{-1})yr(^{-1}))</th>
<th>ES coefficient (2012 US$ha(^{-1})yr(^{-1}))</th>
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</thead>
<tbody>
<tr>
<td>Agriculture</td>
<td>Cropland</td>
<td>92</td>
<td>132</td>
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<tr>
<td>Forest</td>
<td>Temperate/boreal</td>
<td>302</td>
<td>433</td>
</tr>
<tr>
<td>Built-up</td>
<td>Urban</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Grassland</td>
<td>Grass/rangelands</td>
<td>232</td>
<td>333</td>
</tr>
<tr>
<td>Rivers/lakes</td>
<td>Rivers/lakes</td>
<td>8,498</td>
<td>12,193</td>
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Table 5. The area and percentage of each land use/land cover class from 1987 to 2010

<table>
<thead>
<tr>
<th>LULC classes</th>
<th>1987 (ha)</th>
<th>% of total</th>
<th>2010 (ha)</th>
<th>% of total</th>
<th>Net change (ha)</th>
<th>Net change %</th>
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</thead>
<tbody>
<tr>
<td>Agriculture</td>
<td>40,664</td>
<td>33.34</td>
<td>43,288</td>
<td>35.49</td>
<td>2,623</td>
<td>6.45</td>
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<tr>
<td>Forest</td>
<td>45,011</td>
<td>36.91</td>
<td>36,045</td>
<td>29.55</td>
<td>-8,966</td>
<td>-19.92</td>
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<tr>
<td>Built-up</td>
<td>11,370</td>
<td>9.32</td>
<td>14,847</td>
<td>12.17</td>
<td>3,477</td>
<td>30.58</td>
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<tr>
<td>Grassland</td>
<td>24,186</td>
<td>19.83</td>
<td>26,542</td>
<td>21.76</td>
<td>2,356</td>
<td>9.74</td>
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<tr>
<td>Open water</td>
<td>730</td>
<td>0.60</td>
<td>1,240</td>
<td>1.02</td>
<td>510</td>
<td>69.82</td>
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<tr>
<td>Total</td>
<td>121,962</td>
<td>100.00</td>
<td>121,962</td>
<td>100.00</td>
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Table 6. Fourteen major land use/land cover changes from 1987 to 2010

<table>
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<tr>
<th>Period</th>
<th>From class</th>
<th>To class</th>
<th>Area changed (ha)</th>
<th>% of all changes</th>
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<td>1987-2010</td>
<td>Forest</td>
<td>Grassland/shrub</td>
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<td>Open water (Gravel pit)</td>
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<td>Open water (Rivers/lakes)</td>
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<td></td>
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<td>Open water (Gravel pit)</td>
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<td></td>
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<td>Open water (Rivers/lakes)</td>
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<tr>
<td></td>
<td>Agriculture</td>
<td>Grassland/shrub</td>
<td>3342</td>
<td>14.48</td>
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<td></td>
<td>Grassland/shrub</td>
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<td>2047</td>
<td>8.87</td>
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<td></td>
<td>Grassland/shrub</td>
<td>Built-up</td>
<td>63</td>
<td>0.27</td>
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<tr>
<td></td>
<td>Open water</td>
<td>Built-up</td>
<td>32</td>
<td>0.14</td>
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<td></td>
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<td>Agriculture</td>
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<td>Total changed</td>
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Table 7. Ecosystem services value estimated for each land use/land cover class in Butler County using Costanza et al. 1997 ecosystem services coefficients corrected for 2012 US$

<table>
<thead>
<tr>
<th>LULC Classes</th>
<th>1987</th>
<th>% of total</th>
<th>2010</th>
<th>% of total</th>
<th>Net change</th>
<th>Net change</th>
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<tr>
<td></td>
<td>US$ million/yr</td>
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<td>US$ million/yr</td>
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<td>US$ million/yr</td>
<td>%</td>
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<td>13</td>
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<td>Forest</td>
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<td>47</td>
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<td>-20</td>
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<tr>
<td>Built-up</td>
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<td>0</td>
<td>0.0</td>
<td>0</td>
<td>0.0</td>
<td>0</td>
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<tr>
<td>Grassland</td>
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<td>19</td>
<td>8.8</td>
<td>22</td>
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<td>10</td>
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<tr>
<td>Open water</td>
<td>8.9</td>
<td>21</td>
<td>10.8</td>
<td>26</td>
<td>1.9</td>
<td>21</td>
</tr>
<tr>
<td>Total</td>
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<td>100</td>
<td>41.0</td>
<td>100</td>
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Figure 1. Ecosystem services (ES) obtained from Nature (Source: Millennium Ecosystem Assessment Synthesis Report, http://www.maweb.org/en/Products.Synthesis.aspx)
Figure 2. Study Area: Butler County, OH, USA
Figure 3. Example of land use/land cover changes from agriculture to open water: (a) RGB composition image of Landsat TM5 (1987) (b) RGB composition image of Landsat TM5 (2010) (c)-(f) the first four PCs
Figure 4. Land use/land cover change map of Butler County from 1987 to 2010 using PCA-based method
Figure 5. Net area changes from 1987 to 2010 in Butler County, OH (unit: ha)
Figure 6. The percentages of changed areas in each land use/land cover class in Butler County from 1987 to 2010 (unit: %)
Figure 7. Net changes in ecosystem services value estimated for each land use/land cover class in Butler County using Costanza et al. 1997 ecosystem services coefficients corrected for 2012 US$ (unit: million US$yr$^{-1}$)
### Appendix A. Eigenvalues of both 1987 and 2010 PCs

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<tr>
<th></th>
<th>PC1</th>
<th>PC2</th>
<th>PC3</th>
<th>PC4</th>
<th>PC5</th>
<th>PC6</th>
<th>PC7</th>
<th>PC8</th>
<th>PC9</th>
<th>PC10</th>
<th>PC11</th>
<th>PC12</th>
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<tr>
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65
Appendix B. PC1 (79%)
Appendix C. PC2 (10%)
Appendix D. PC3 (6%)
Appendix E. PC4 (2%)
Appendix F. Aerial photos showed a couple of mining operations that were classified as open water, which have ES value as zero US$