Put a Price on It: Measuring Ecological Value of a Diverse Landscape

Kristie Sills, Becca Halter, Andy Frank
Environmental Studies and Sciences
Skidmore College
Spring 2017

Abstract
This research is a case-study measuring the value of ecosystem services in a 138 hectare parcel of Skidmore's undeveloped land. We estimate the parcel's annual value is $150,896. The market value of the parcel's carbon sequestration (regulating service) is $164. The value of mountain biking (cultural service) through a revealed preference survey of the Saratoga Mountain Bikers Association (n=19) is $676. Surveys of Skidmore students (n=162) assessing the stated-preference value of the parcel is $3,219. We used Wetland Ecosystem Service Protocols and field based carbon storage measurements. In our analysis, we found most carbon is stored in the soils (~50%) and wetlands have the highest ecosystem value per hectare, but their ecological condition and risk varied. We suggest Skidmore College should include ecosystem services in their land management plan.

Keywords
Ecosystem services, market and non-market value, stated preference, revealed preference carbon storage, WESP, InVEST

Introduction

The Need for Ecosystem Services

With increasing demand on natural land, ecological economists have established techniques to ascribe dollar values to the benefits ecosystems provide to humans. The effort to measure the goods and services ecosystems provide to humans is a salient way to address systemic and synergistic environmental change. In technical terms, the "goods and services ecosystems provide to humans" simply refer to "ecosystem services." Failure to account for the full economic value of ecosystems and biodiversity are the most significant factors in the loss and degradation of ecosystem services (TEEB, 2010). Considering 60% of the world's ecosystem services are being degraded or used unsustainably, current value estimations fail to represent the full economic value of the goods and services ecosystems provide (MA, 2005). Present ecosystem degradation, and future ecosystem stressors necessitate markets to regulate the consumption and conservation of the environment.

Human dependency on environmental systems is the impetus of ecosystem valuation. The production and provision of goods, development of society, and functioning of the planet's systems are a compilation of the world's capital stocks. Manufactured capital, human capital, and natural capital generate a flow of services that enhance human welfare. Manufactured, or built capital, enable the production and provision of goods through machines, tools, technology, and infrastructure. Human capital or the skills, knowledge, and experience of laborers create development. Natural capital, or the planet's ecosystems and natural assets, sustain the functioning of the Earth's life support system. Natural capital, is abundant, easily exploitable, and serves as raw material for human development. The natural world is at the crux of the money economy, providing the basis of human welfare.

Currently, several of the Earth's global, regional, and local environmental thresholds are close, or have been exceeded (Rind, 2008). Once such thresholds have been exceeded, abrupt and possibly irreversible change will inhibit the planet's life support systems. This prompts the question, "Are Earth's critical thresholds being strained because humans are not attaching proper economic value to them?" Which brings about the question, "Have traditional economic studies adequately valued human welfare loss due to ecosystem change?"

Traditionally, economic studies measure welfare loss due to climate change according to the Intergovernmental Panel on Climate Change temperature projection of a 2.5°C temperature increase. Using a 2.5°C temperature increase, (Nordhaus, 1994; Fankhauser, 1997; Tol, 1996) predict the cost of climate change is equivalent to the loss of one year's growth in the global economy. However, increasing certainty of abrupt changes in the climate system (Alley et al., 2003; Lenton et al., 2008) indicate that the assumption the climate will change smoothly is likely false. A growing number of economists contend that the assumption of a smoothly changing planet significantly underestimates the costs of abrupt ecosystem change (Stern, 2006; Lemoine and Traeger, 2012; Cai et al., 2015). Modern ecosystem valuation builds upon the integration of natural capital into economic and societal systems, and is most heavily founded upon the

Millennium Ecosystem Assessment (MA). The great problem facing ecosystem valuation is the environment's lack of an exchange value. Due to the evolution of classical and neoclassical economic theory, economics involves the study of scarcity, the production of commodities, and distribution of goods. Goods and services that are sold as commodities in markets have an exchange value. An exchange value allows a good or service to have a monetary unit of measurement. Since ecosystems lack an exchange value the market fails to regulate its sustainable use. This has led to the so-called "invisibility" of environmental services from economic systems. The ultimate purpose of ecosystem services is to use economic values to make the non-market benefits of the environment more visible.

Estimates of the total economic value of ecosystem services (Costanza et al., 1997; De Groot et al., 2012; Costanza et al., 2014) garnered widespread attention to the monetization of ecological processes. However, such aggregates are a poor reference to local accounts of ecological value. While there are many ways to quantify ecosystem services for given areas, global estimates and two-thirds of all of the literature rely on satellite imagery and/or land cover data sets that reflect entire regions rather than localized ecosystems (Seppelt et al., 2011). Less than 40% of all studies done to quantify ecosystem services include primary data and field measurements (Seppelt et al., 2011). Global estimates lack in-the-field biophysical assessments for specific land values, and therefore do not accurately describe how ecosystems impact people on a local scale. Ecosystems are dynamic and unique, and provide their services very differently depending on location. While many methods exist to quantify ecosystem services globally, there is no uniformly accepted method to quantify ecosystem services on a local scale (Hein, 2014).

Localized assessments of ecosystem services offer the most methodological consistency by accounting for ecosystem heterogeneity. The assessment of a large land parcel, as opposed to the planet or vast regions, makes the associated dollar value more digestible to individuals involved in the study who may be unfamiliar with the concept of ecosystem services. It is impossible to authenticate if the world's ecosystem services and natural capital are worth $125 trillion USD/ha/yr. However, localized assessments of ecosystem services offer insight into the accuracy of global estimates of ecosystem services. Furthermore, local assessments can help inform ecosystem services as a broader accepted policy.

Purpose of this Research

The first step to value ecosystem services is to establish the environmental baseline by determining the location and type of ecosystem services present in a given area (Price, 2007).

The measurement of ecosystem services utilizes quantitative (ecosystem service flows), spatial (mapping), and qualitative indicators (stakeholder perceptions and preferences) to illustrate the value of ecological processes (De Groot et al., 2012). Thus, the measurement of ecosystem value requires a mixed method approach incorporating quantitative, spatial, and qualitative assessments. Our study uses the framework of ecosystem services defined by the Millennium Ecosystem Assessment to assess the ecological value of a small parcel of land. This research includes ecosystem service estimates according to quantitative field measurements of forest carbon storage, Regional Greenhouse Gas Initiative (RGGI), Wetland Ecosystem Services Protocol (WESP), and Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST). We also employ revealed and stated preference techniques to estimate the dollar value of ecosystem services by surveying two stakeholder groups. In addition to our market and non-market estimations of ecological value, our survey also addresses key issues in the literature of ecosystem services by identifying how people perceive the economic conceptualization of the environment. Understanding the extent people interpret the economic contribution of the environment is dependent on an accurate rendering of value. Differentiating between methodologies helps clarify how people perceive environmental value. Assessing how people relate to the environment is an important step to standardizing ecosystem services as a measure of environmental value.

Literature Review

Relevant Economic Theory

The history of ecosystem services in economic theory elapses over three principal schools of economic thought. Classical, neoclassical, and ecological economic theory shape the economic conceptualization of ecosystem services. During classical economic theory in the late 18th century, the environment was understood according to labor theory and considered a production input generating rent (Gómez-Baggethun et al., 2010). Classical economic theorists' recognition of land as an input to production and determinant of output demonstrates significant environmental awareness considering ecology was not termed until 1866. However, the goods and services provided by the environment lacked an exchange value and were subsequently considered "free." By the end of classical economic theory in 1870, only commodities with an exchange value were relevant to the industrial growth, technological development, and capital accumulation of the 19th century (Gómez-Baggethun et al., 2010). By the 20th century, neoclassical economic theorists exclude land as an input of production (Gómez-Baggethun et al., 2010). The allowance for capital to replace land in the mid-20th century lead neoclassical economists to consider the environment a substitutable input. Since capital could substitute land, the regulation of the environment was considered unnecessary. The effective removal of land from the production function made the environment obsolete to measures of national wealth.

The early omission of ecological processes from calculations of economic value lead to the systematic lack of markets for environmental goods and services. Without markets, or prices, the value of ecological functions that underlie human processes are invisible to the economy. Currently, the functions of ecological processes do not have prices. Prices control markets and influence the actions of agents. Markets with desirable conditions will result in environmental degradation; agents of environmental degradation chose to degrade because it is profitable (TEEB, 2010). The purpose of pricing environmental services is to ensure actions leading to environmental degradation are not profitable.

Recognition of environmental degradation in the 1950's lead economists to reevaluate the influence of economic systems over environmental systems. The subsequent fields of resource economics, environmental economics, and ecological economics, all seek to eliminate the segregation of the environment from the economy. The field of ecological economics is unique to its counterparts of resource economics and environmental economics with its understanding of the economic system as a subsystem of the ecosphere (Daily, 1997). The concept of strong sustainability, or the rejection of manufactured capital as a substitution for natural capital, is also exclusive to ecological economics (Daly, 1992). The academic discipline of ecological economics was first institutionalized in 1989 by the Journal of Ecological Economics (Røpke, 2004). Since its inception, the journal has sought to integrate "humankind's household" with "nature's household" by emphasizing the interdependency of social and ecological systems. Modern ecosystem valuation builds upon the integration of natural capital into social and economic systems, and is most heavily founded upon the MA. According to the findings of the MA, 60% of the world's ecosystem services are being degraded or used unsustainably.

Market and Non-Market Value Elicitation

The historic exclusion of the environment from economic systems has led to public policy and government investments that exacerbate the degradation of soil, air, water, and biological resources. Human dependency on environmental systems is the impetus of ecosystem valuation. Biological systems sustain human activities, including the energy humans use for transport, the raw materials needed to produce goods, the provision of food, and the disposal of waste (TEEB, 2010). MA (2005) and TEEB (2010) employ a framework of total economic value (TEV) to measure the *use value* and *non-use value* of an ecosystem. An ecosystem's use value is based on how the individual benefits from the environment, while non-use value accounts for environmental benefits external to the individual. These two categories, use and non-use value, encompass the market and non-market benefits of ecosystem services (TEEB, 2010).

There are three categories of an ecosystem's use value; direct use, indirect use, and option value (MA, 2005). Direct use includes, extractive, consumptive, and nonconsumptive benefits. For example, the extraction of timber, consumption of food, and nonconsumptive benefit of recreation, all reflect the direct use of ecosystem services (De Groot et al., 2012). Indirect use values represent the benefits rendered by ecosystems without direct use by the individual. Regulating and supporting services indirectly support humans, by providing basic ecological services that maintain ecosystem functionality. For example, water filtration in a wetland provides drinking water for an individual. The individual is not directly using a wetland, but still receives the benefits of water filtration. Lastly, an option value represents the future use of a resource even if the individual is not a current user.

There are also three categories of an ecosystem's non-use value including the existence value, altruistic value, and bequest value (MA, 2005). The conservation of an endangered species is an example of existence value. The individual values a natural resource for the sake of its existence. The altruistic and bequest value represent the value of a natural resource for people other than the individual. The altruistic value is an ecosystem's availability to the current generation and the bequest value is the availability of ecosystems to future generations.

Total economic value estimations of ecosystem services rely on both market and non-market appraisals of environmental value. Most market schemes for ecosystem services account for carbon sequestration in biomass or soil, provision of habitat for endangered species, protection of landscapes, and the preservation of the quality and quantity of hydrological functions (Gómez-Baggethun et al., 2010). For example, the 1972 Clean Water Act established the market value of the hydrological functions supplied by wetlands. As a result of the bill, the Corps of Engineers issue permits to allow damage to wetlands in exchange for the creation or restoration of larger wetlands elsewhere (Gómez-Baggethun et al., 2010). According to the Corps of Engineers, the average cost of wetland mitigation per acre is $45,000 (Pandeya et al., 2016). The cost of building a constructed wetland to mimic the water filtration of a natural wetland is an example of the replacement cost (Pandeya et al., 2016). The replacement cost quotes the price of replacing ecosystem services. Some ecologists assert the replacement cost only accounts for a single ecosystem service and subsequently undervalues the ecological processes of the entire ecosystem (Winpenny, 1991).

Non-market ecosystem services are harder to appraise than their market counterparts. Unlike market systems, there is no formalized methodology of the non-market value of ecosystem services. Ecosystem services that lack market prices rely on stakeholder participation to express the revealed or stated preferences of an individual or group. Revealed preference and stated preference estimations ascertain value to otherwise non-market goods. Revealed preference relies on existing markets to determine value while stated preference utilizes hypothetical markets.

In revealed preference methods, like the travel cost and hedonic pricing, the value of an ecosystem is revealed through the consumer's behavior in established markets. Revealed preference methods determine ecosystem value according to consumer spending habits (Pandeya et al., 2016). Both travel cost and hedonic pricing utilize values external to the functions of the ecosystems. Hedonic pricing appraises changes in environmental quality according to changes in housing prices by using real estate markets serve as a proxy to valuing environmental change (Pandeya et al., 2016). The travel cost method estimates the value of recreation, a non-market environmental good, according to the amount of time and travel expenses people incur to access a given site (Brown & Mendelsohn, 1984). The annual price of a given recreation area is calculated according to the annual number of trips and distance commuters travel to get to a specific site (Brown & Mendelsohn, 1984). A fundamental problem of revealed preference is the reported lack of relationship between spending behavior and the value of non-market ecosystem services (Costanza et al. 1997).

Hypothetical markets created in the stated preference method, sometimes referred to as Contingent Valuation, elicit an individual's willingness to pay (WTP) for an increase in the quality of an environmental condition. Alternatively, stated preference may assess how much an individual is willing to accept (WTA) for a decrease in environmental quality (Bateman et al., 2002). Another variation of Stated Preference includes choice modeling studies which examine the choices individuals make under experimental conditions to estimate the value of an environmental good or service (Pandeya et al., 2016). The individual is asked to state a preference between one group of environmental attributes at a given cost and another group of environmental attributes at a different cost. Choice modeling is most applicable to policy decisions where a set of possible actions will have differing environmental impacts. One advantage of the stated preference method over revealed preference method is that stated preference is able to explicitly estimate the value of a specific ecosystem service while revealed preference only applies to ecosystem services that involve consumer spending. Opponents of the Stated Preference method repudiate the measure of WTP or WTA, criticizing the technique as an attempt to measure ethical and moral questions rather than representing the value of the ecosystem services (Zander et al., 2014).

In addition to market based approaches, revealed preference and stated preference, economists utilize benefit value transfer to estimate TEV. Benefit value transfer is controversial because it uses empirical value estimates obtained from market based approaches, revealed preference, and stated preference to extrapolate the value of ecosystem services on regional or global scales. Aggregate valuation of ecosystem services is methodologically challenging because there are both communal and individual benefits provided by ecosystems.

Localized valuation of ecosystems is preferable to aggregate value because the communal and individual benefits of a given service are more readily distinguished and separated (Farber et al., 2002). Local ecosystem assessments focus on a single service and generally include an ecological production function (Price, 2007). Economic values elicited from ecological production functions characterize the relationships between ecosystem condition, management practices, and the provision of ecosystem services (Macpherson et al., 2010). However, ambiguity between economic modeling and ecological functions complicate the design of ecological production functions (Macpherson et al., 2010). Instead of a single framework, concept pluralism is necessary to understand the intersection between ecological value and economic systems in order to accommodate the different types of services provided by the natural environment.

Framework of Ecosystem Services

The Millennium Ecosystem Assessment (MA) is the first international study to discern the market and non-market benefits ecosystems provide to human well-being. The resulting framework of "ecosystem services" is a UNEP collaboration of 95 countries, and 1360 contributors, initiated by the Convention on Biological Diversity, UN Convention to Combat Desertification, Ramsar Convention on Wetlands, and the Convention on Migratory Species.The report was a joint request for a scientific assessment of global ecosystems. In 2005, the four-year analysis was complete, establishing a scientific basis with the aim of increasing the sustainable use of ecosystems.

The MA provides the foremost authority informing the valuation of ecosystem services. The MA defines ecosystem services as the benefits people obtain from ecosystems (MA, 2005). Such benefits include *provisioning services* providing food, water, timber, fuel, and fiber; *regulating services* affecting climate, floods, disease, waste, and water quality; *cultural services* supplying recreation, education, aesthetic, and spiritual benefits; and *supporting services*, that are necessary for the production of all other ecosystem services and include processes like soil formation, photosynthesis, and nutrient cycling. Value appraisals for provisioning services and cultural services are well established in the literature. Market prices for food, water, timber, fuel, and fiber, readily transmit the value of provisioning services (TEEB, 2010). Additionally, the direct benefits of cultural services, such as recreation and appreciation of nature, are easily observed and understood (Price, 2007). In contrast, value appraisals for the indirect benefits of regulatory services and supporting services are largely unknown and ill established in the literature (Price, 2007). However, global assessments of ecosystem value associate the regulating and supporting services of forests and wetlands with the highest dollar amounts amongst all terrestrial ecosystem services (Costanza et al., 1997; De Groot et al., 2012). The values of regulating and supporting services are less well known because the benefits they provide are outside the market and consequently referred to as non-market benefits. For example, the benefits of flood regulation (regulating service) and nutrient cycling (supporting service) that occur in wetlands and forests are not traded in markets.

Without markets, there are no prices, and ultimately, no representation of the economic value of flood regulation and nutrient cycling. The high ecological value of the regulatory and supporting services of a forest and wetland are at odds with their respective non-existent economic value. The Economics of Ecosystems and Biodiversity (TEEB), the second utmost authority on ecosystem services, has sought to ease the tension between economic value and ecological value by demonstrating how ecosystem services impact economic activity and human well being. The environmental ministries of G8 + 5 (Canada, France, Germany, Italy, Japan, the United Kingdom, the United States, and Russia) organized TEEB in 2007 to act as a catalyst for global economies to incorporate assessments of ecosystem services into mainstream decision making.

One such opportunity to utilize the framework of ecosystem services into decision making, is land management plans. Generally, the first component of a land management plan is a declaration of intent establishing the aim of its creation. The aim of the land management plan and often include, raising awareness of ecosystem services, establishing multiple-use goals and objectives, and promoting ecological integrity (Jenkins et al., 2003; Zadrozny Brenner, 2011; Alexander et al., 2014). The extent land management plans influence daily decision making is unclear. However, the inclusion of ecosystem services in a land management plan is a progressive commitment to legitimizing measurements of ecological value.

The discipline of ecological economics examines the value of ecosystem services, the relationship between economic growth and the environment, the effect of production and consumption on environmental quality, and the exploitation of public goods (Costanza et al. 2004). The majority of the studies published in the field of ecological economics are valuation studies of ecosystem services (Costanza et al. 2004; Ma & Stern 2006; Chaudhary et al., 2015) The literature concerning ecosystem services is growing exponentially, over 5,000 reports were published between 2010 and 2013 in contrast to the 326 reports that were published between 1997 and 2000 (Chaudhary et al., 2015). The publications by (Daily, 1997; Costanza et al., 1997; MA, 2005; TEEB, 2010) are the most influential standards of ecosystem service valuation. The definition of ecosystem services by (Daily, 1997) require economic analysis to represent a synthesis of the biosphere:

> "*Ecosystem services are generated by a complex of natural systems driven by solar energy that constitute the workings of the biosphere*" (Daily, 1997, p. 4).

The attempt at such a synthesis by (Costanza et al., 1997) was the first comprehensive economic estimate of the world's ecosystem services. The definition provided by (Daily, 1997) is congruent to the classification of ecosystem functions by (Costanza et al., 1997). The authors (Costanza et al., 1997) categorize 17 ecosystem services according to the ecosystem function of various biological processes using benefit value transfer to ascribe units to marine[1] and terrestrial systems.[2] The unit value of a given ecosystem was provided in dollars per hectare per year ($/ha/yr). The average annual value of global ecosystem services was $33 trillion USD/ha/yr (in 1995 $US). At the time of the study, gross national product in the United States was $18 trillion USD.This value has been cited over 17,000 times, proliferating the literature of ecological economics as the most cited dollar estimate of ecosystem services (Costanza et al. 2004; Ma & Stern, 2006). Despite its prevalence in the literature, there is widespread criticism of the study's use of benefit value transfer. The methodology is controversial because it applies previous empirical estimates of ecosystem value to similar environments. As a result of benefit value transfer, ecosystem services are given equal value, regardless of biophysical and spatial variability. The extrapolation of value from a given ecosystem to a global scale neglects heterogeneity amongst ecosystems (Turner et al., 2003; Hein, 2014; Pandeya et al., 2016). The omission of value heterogeneity decreases the accuracy of ecosystem value estimations derived from benefit value transfer.

The authors of the 1997 landmark study concede benefit value transfer is a crude approximation, but emphasize the method is consistent with the type of calculations that produce the gross domestic product. Just as benefit value transfer aggregates ecosystem value, GDP aggregates value by multiplying the price and quantity of each sector of the economy (Costanza

[1] Marine systems included open and coastal ocean, estuaries, seagrass, algae beds, coral reefs, and shelf systems

[2] Terrestrial systems included tropical, temperate, and boreal forests, grasslands, rangelands, wetlands, lakes, rivers, desert, tundra, ice/rock, cropland, and urban environments.

et al., 2014). The purpose of global aggregates of ecosystem value is to inform the contribution of ecosystem services to human well-being relative to national aggregates of welfare like GDP.

A secondary assessment of global ecosystem value using benefit value transfer by (De Groot et al., 2012) refrain from reporting a cumulative value and instead list the average annual hectare value of a given ecosystem ranging between a minimum of $100 USD/yr and a maximum of $1,000,000 USD/yr. The authors credit the significant range to variation in methodology, study sites, socioeconomic conditions, biomes, and ecosystem services. The authors admit benefit value is problematic and state their study does not “underplay the shortcomings and limitations of monetary valuation” (De Groot et al., 2012, p. 56). However, despite the acknowledged limitations of monetization, the authors assert monetization is necessary to convey the gains and losses associated with ecosystem restoration and degradation.

The most recent assessment on the global value of ecosystem services is a follow up to the original 1997 study. The update, by the same authors, accounting for changes in unit value and ecosystem functionality report the annual value of the world’s ecosystem services in 2011 was $125 trillion USD/ha/yr (Costanza et al., 2014). The authors regard the update as conservative and attribute most of the increase in unit value to more ecosystem service value estimations available in 2011 than 1997. This means that better understanding of science, and more research indicate that there is more value in ecosystem services than the value previously stated in 1997. This change in value, without any substantial evidence of quantity or quality changes demonstrates ecosystem services are becoming more valuable.

Despite the author's emphasis on conservation and the value of ecological processes, many economists and ecologists have rebuked their estimate of total economic value. Opponents argue multiplying the value of a hectare per biome neglects the influence of scarcity on value. A general extrapolation of the average value of an ecosystem service does not represent how the value of an increasingly scarce commodity increases (Masood & Garwin, 1998). The authors calculations have also been labeled reductionistic and a “a serious underestimate of infinity” (Toman, 1998, pg. 58). Since the value of all ecological processes is unknown, (Vatn and Bromley, 1993) assert the value of ecological processes will remain unknown until ecosystems stop functioning. The arguments against (Costanza et al., 1997) are numerous and varied, most critics acknowledge the study is responsible for expanding the amount of attention given to ecological processes.

Broad criticism of the discipline of ecological economics assert understanding the benefits of ecosystems according to the benefits they provide to humans disregard the intrinsic value of nature (Vira & Adams, 2009; Peterson et al., 2010; Schröter et al., 2014). Critics argue the anthropocentric framing of ecological processes results in the confusion of ecosystem functions and ecosystem services (Chaudhary et al., 2015). The yield of ecosystem services depends on a variety of biological and physical characteristics. If the characteristics of an ecosystem diminish the output of an ecosystem service the value of the ecosystem function decreases despite its unchanged importance to the overall ecosystem (Vira & Adams, 2009). Placing more emphasis on ecosystem functions rather than the product of ecosystem services allows for greater inclusion of abiotic and biotic factors indicative of ecosystem health (Peterson et al., 2010). Critics also emphasize economic measures of ecosystem value are not necessarily representative of ecosystem functionality. Monetization reduces ecological function into an ecosystem service and ignores the status of the ecosystem (Kosoy & Corbera, 2010). For example, a forest with limited biodiversity and invasive species can still deliver the ecosystem service of carbon sequestration. However, the counterargument by (MA, 2005; TEEB, 2010) contend biodiversity increases the provision of ecosystem services. The dependence of ecosystem services on biodiversity consequently prioritizes the importance of ecosystem functionality.

Skidmore College Land Management

Skidmore College's Sustainability Plan for 2025 outlines five specific sustainability focus areas for the college to achieve. In particular, our assessment addresses three of the five focus areas; Energy, Lands and Grounds, and Engagement. Lands and Grounds, focus area four of Skidmore's Campus Sustainability Plan for 2025, requires that Skidmore create comprehensive land management plans for all of its land parcels. Skidmore College has over 1,000 acres of developed and undeveloped lands, and currently only the 150 acre Northwoods parcel has any form of management with a trail map, signage, and Northwoods Stewards (Skidmore Woodland Atlas, 2012). This research project explores the value of ecosystem services in one of Skidmore's undeveloped parcels adjacent to Daniels Road State Forest. By putting a price on ecological value, we explore the way ecosystem service assessments can inform Skidmore's land management. With the suggestion for Skidmore's land management plans, our study also provides solutions to Skidmore's goal of Engagement, or focus area five, of the Campus Sustainability Plan.

Additionally, Skidmore's Campus Sustainability Plan for 2025 outlines the use of their forested land as a carbon sink to offset the college's greenhouse gas emissions (Campus Sustainability Subcommittee, 2015). Acknowledging the fact that GHG mitigation is critical in the face of climate change, it is also crucial to understand that the value of natural land goes far beyond carbon sequestration. Ecological services and the broader ecological value of Skidmore's land needs to be assessed in order to develop a comprehensive and innovative management plan that is suitable to Skidmore's future sustainability goals. As scientifically informed management plans become more common, it is critical for institutions like Skidmore College to become a leader in implementing land management plans based on a prioritized assessment of ecosystem services.

Many small institutions similar to Skidmore possess natural land parcels and allocate resources towards their general conservation. Institutions such as Hamilton, Vassar, Kenyon, Bard, Connecticut College, Franklin and Marshall, Dickinson, St. Lawrence, and Middlebury all possess forested land parcels between 5-800 hectares (ha) (Freeman et al. 2012). While these institutions use their forested land for research and education, none of them have specifically included the ecosystem services provided by the forested lands as reason for conservation. There is a lack of research into ecosystem services by institutions similar to and including Skidmore, and this gap in the connection between ecosystem services and land conservation gives Skidmore an opportunity to set a precedent for future conservation efforts.

In 2010, Skidmore College was donated a 200 acre (81 ha) parcel of land located to the north of the college's main campus. The land was donated by the owners of Roohan Realty, a local real estate company in Saratoga Springs, NY. By 2016, an additional donation of 140 acres (57 ha) to the north increased the overall size of this parcel to 340 acres (138 ha). The parcel is naturally forested with three distinct wetlands within it. Part of Loughberry Lake's watershed lies adjacent and partially within the parcel (Figure A4), meaning this parcel has potential to impact the future health of Loughberry Lake, which is Saratoga Springs' drinking water source.

To examine how ecosystem services can be valued and assessed on a local scale, we used this parcel as a case study and measured the services provided by its forests and wetlands. Not only does our assessment establish a framework for small institutions like Skidmore to begin to consider ecosystem services within their own economic and environmental agendas, but it also directly helps Skidmore in moving towards accomplishing three of its five focus areas for Sustainability by 2025.

Methods

Study Site

Our study is focused on a new addition to Skidmore's property assets located in the Town of Greenfield, NY (Figure 1). The property was a generous gift to Skidmore courtesy of the Roohan family of Roohan Realty. The land was donated without outlined intentions within the deed, but rather with the hope that Skidmore will conserve the land to conduct research and further educational opportunities (Freeman et al., 2012).

Figure 1. *Map of the study parcel and surrounding area. Satellite image obtained through NYS orthoimagery.*

The parcel has a total area of 138 ha, of which roughly 105 ha is primarily mixed hardwood and conifer forest dominated by second-growth *Acer saccharum*, *Quercus rubra*, *Pinus strobus*, *Betula allegheniensis,* and *Tsuga Canadensis*. The remaining area of about 32 ha consists of peat-forming wetlands that are a mix of open sedge (*Carex spp.* dominated), wooded (*Picea spp.* dominated) riverine fens with seasonal water, and open-water wetlands.

Bedrock in the area is variable and consists of crystalline rocks, sandstone, carbonate rocks, and shale. Ice retreated from the Mohawk-Hudson lowlands 10,000 years ago causing temporary glacial lakes to form and depositing till across the region. Our study parcel lies on a ridge consisting east-west lengths of bedrock outcrops and superficial glacial till and swamp deposits (Heisig, 1994). The soils are rocky with a thin organic layer and an underlying fine sandy loam (Charfield-Hollis complex; Soil Survey Staff, n.d.).

We used geographic information systems (GIS) to create maps of Skidmore's parcel, as well as a random point generator to obtain 30 points within the 105 ha forested area. These points were used for our carbon storage measurements of the forest. We accessed the parcel through Hilltop Dr., an access road directly off of Route 9. In total, our in-the-field quantitative assessment required 110 km of walking and surveying the landscape (Figure A3).

Population

This study surveyed members of the Skidmore College community including the City of Saratoga Springs and the Saratoga Mountain Biking Association (Table 1).

Relevant Stakeholders	Population
Skidmore College	2,450 Skidmore College students 259 full time faculty members
Saratoga Mountain Biking Association	1,070 social media followers

Table 1. *Stakeholder Population.*

The population demographics of the City of Saratoga Springs demonstrate an affluent, predominantly white community (Table 2). Archival research from "Admissions Facts" describe the Skidmore College student body as 22% domestic students of color, 10% international students, 5% students with dual passports, and 13% first-generation college students.

	Median income	Identify as white
Saratoga Springs	$70,187	92.3%

Table 2. *Population Demographics of Saratoga Springs. Data source from the U.S. Census Bureau (2015).*

We choose to survey these two populations because students make up the bulk of the Skidmore College population and the Saratoga Mountain Biking Association (SMBA) is the biggest user of the land adjacent to our survey site. Some of the mountain biking trails SMBA built overlap onto the Skidmore parcel (Figure A5). While we choose to focus on Skidmore students and SMBA, this parcel is relevant to the greater Skidmore College community conservation effort in Saratoga County. The parcel is implicated in an area that concerns many landowners, developers, researchers, hunters, and residents. Interested stakeholders include Skidmore College Sustainability Office, Saratoga PLAN, Saratoga Mountain Bike Association, City of Saratoga Springs Chamber of Commerce, Moreau Lake State Park, Saratoga Spa State Park, and Wilton Wildlife Preserve and Park among many others.

Survey Methodology

Comprehensive qualitative analysis uses triangulation to verify data by cross referencing two or more sources (Creswell, 2012). The use of three methods, market based approach, stated preference, and revealed preference triangulate our qualitative estimate surveying ecological value. The benefits ecosystems provide to humans that lack prices to represent their value are the “non-market benefit” of an ecosystem and are measured through behavioral patterns and hypothetical markets. Behavioral patterns utilize established markets to reveal people's environmental preferences. This methodology is known as revealed preference. Hypothetical markets estimate how much people are willing to pay for a given environmental good or service. This method is known as stated preference. We surveyed perceptions of ecological value from two different stakeholder groups, Skidmore college students (n = 162), and the SMBA (n = 18). Two separate surveys were used to measure student perception of ecological value and SMBA perception of recreational value (Appendix D).

The survey evaluating student perception of ecological value also quizzed students' knowledge on the general definition of ecosystem services and more specific wetland and forest ecosystem processes. Students were asked 6 questions about different wetland characteristics and functions, 3 questions about forest processes, and 2 questions about the definition of ecosystem services. The quiz was used to address the following questions: To what extent do people understand the relationship between ecosystem services and ecosystem functions? Does prior knowledge of ecosystem functions influence the way people value ecosystem services? After assessing the extent students understand ecosystem services, we sought to gauge how students perceive ecological value. The survey distributed to students provides our Stated Preference estimate of ecological value by asking students how much of their tuition they would be willing to allocate toward ecosystem services. This method of soliciting someone’s willingness to pay is controversial, but widely used in the literature.

The survey distributed to SMBA is the basis of our revealed preference estimate of ecological value. We adapted the Travel Cost Method to estimate recreational value of a nearby state owned parcel (Brown & Mendelsohn, 1984). SMBA members were asked how many trips they made annually to the Daniels Road State Forest trailhead, how many miles they drove to get to the trailhead, and the amount they spend annually on bike maintenance. The cost of gas was calculated using the Skidmore Student Government Association metric for gas reimbursement as set by the College at .35 cents per mile. The number of trips was then multiplied by the cost of gas and added to bike maintenance to calculate an estimate of the yearly expenditure SMBA members spend on mountain biking. We used Daniels Road State Forest as a proxy to our parcel because the area is geographically similar to our area of study.

Carbon Storage

We surveyed the northern most parcel owned by Skidmore College with the goal of obtaining a sampling intensity of 2%. We preselected 30 randomly distributed 15 m radius plots throughout the parcel using the ArcGIS random sample calculator tool. Each plot was located in the field using a Magellan GPS unit and marked with PVC pipe. Plots where the edge overlapped wetlands were shifted on site to include only upland forest cover. Each plot was additionally classified by their land cover type using their geographic location and the matching land cover from the NOAA's Coastal Change Analysis Program (C-CAP) 2010 Land Cover Classification data (NOAA, 2010).

To calculate the aboveground carbon (AGC) and belowground carbon (BGC), we estimated the aboveground biomass (AGB). AGB was obtained by measuring the diameter at breast height (DBH) of all trees >2.5 cm diameter within our sampling plots. AGB for each tree, based on their species group, was then calculated using the equation:

$$AGB = Exp(\beta_0 + \beta_1 ln(dbh))$$

where β_0 and β_1 are parameters for each species group (Table 3, Jenkins et al. 2003). AGC was calculated from the AGB under the assumption that AGC = AGB x 0.47 (Hughes et al. 1999). AGC includes aboveground foliage, bark, stem wood, stem bark, and branches. Any unknown species were classified in the Pine group for calculations because the Pine values were the smallest for the species groups in the Northeast and therefore give an underestimate. BGC was estimated with the equation:

$$BGBD = e^{1.059 + 0.884 \times ln(AGBD) + 0.284}$$

where AGBD is the aboveground biomass density calculated from the previous section divided by the plot area (Cairns et al., 1997 as seen in Brown et al. 1999). First the AGB density (AGBD) was calculated from the AGB data. Both AGBD and BGBD are calculated in Mg ha^{-1}. This equation includes all living and dead roots, and the studies this equation is founded on are unclear about what root size classes and depths that were examined (Cairns et al., 1997). We used the same assumption as AGC (Hughes et al., 1999), that 47% of the biomass is carbon:

$$BGC = BGC \times 0.47$$

		Parameters	
	Species Group	**β0**	**β1**
Hardwood	Aspen	-2.2049	2.3867
	Soft maple / birch	-1.9123	2.3651
	Mixed hardwood	-2.48	2.4835
	Hard maple, oak, hickory, beech	-2.0127	2.4342
Softwood	True fir / hemlock	-2.5384	2.4814
	Pine (unknown species)	-2.5356	2.4349

Table 3. *Aboveground biomass calculation parameters adapted from Jenkins et al. 2003.* $AGB(kg) = Exp(\beta_0 + \beta_1 ln(dbh))$. *DBH was measured in cm, and only trees > 2.5 cm diameter were included. Unknown tree species were classified under the Pine parameters, as these values give the smallest value for any species group found in a Northeastern forest.*

At each of our plots we randomly selected two 20 cm by 20 cm squares to measure the carbon in the leaf litter and forest floor. The leaf litter and forest floors were collected separately. We air dried and then weighed each sample. The leaf litter mass was assumed to be 47.0% C and the forest floor 37.6 % C (Wind, 2013; Smith et al., 2006). Our study does not include an estimate for any standing or downed-dead wood. Using sample areas of the AGC, BGC, leaf litter, and forest floor measurements, we calculated the carbon stored in each component per hectare to get carbon storage for all components in Mg C ha^{-1}.

At each of our plots, we took two soil cores using a beveled 1.5 inch PVC pipe. Soil cores were composited for a 0 to 10 cm and a 10 to 20 cm soil profile. Composite samples were sieved to 2 mm, dried at 105°C for 24-30 hours (Fisher Isotemp Oven Model 350, Fisher Scientific, Pittsburgh, PA), and then ground-up using a ball mill (SPEX SamplePrep, Metuchen, NJ USA). Each sample was then dried again at 75°C for at least 12 hours to remove excess moisture. Approximately 20 mg of each sample was then weighed and wrapped in 5 x 9 mm tins and combusted in an ECS 5010 elemental analyzer (Costch Analytical, Valencia, CA, USA) to estimate total C content. A bulk density (⅓ Barr) of 1.01 g cm^{-3} for the 0 - 10 cm samples and 1.38 g cm^{-3} 10 - 20 cm samples was used to calculate the soil carbon density and the sample depths gave us the soil carbon content per m^2 (Soil Survey Staff, n.d.). Using our plot areas, we calculated the soil carbon stored in each component per hectare.

Wetland peats were cored at each of the wetlands using a Standard Russian Peat Borer (Aquatic Research Instruments, Hope, ID). Peat sections in 10 cm lengths were dried, ground up, and analyzed in the same manner as the soil samples. Bulk densities were measured using their initial weight divided by the volume of the core:

$$bulk\ density = \frac{mass}{volume} = \frac{sample\ weight\ (g)}{10cm \times \pi \times (4.6cm)^2}$$

Using a zinc C-rod, we measured the depth of the peat in multiple spots across each wetland to get an estimated average depth of each wetland. Given the bulk density and peat depth, we estimated the carbon stored per hectare in each wetland type.

Finally, based on each plot's 2010 C-CAP land cover type we extrapolated each components carbon storage density (Mg C ha^{-1}) to achieve parcel wide estimates (Mg C).

Wetland Ecosystem Service Protocol

We used the Wetland Ecosystem Services Protocol (WESP) test version 1.3 to determine a rapid assessment of ecosystem service values for the wetlands within the parcel (Adamus, 2016). WESP is a model designed to evaluate multiple wetlands over a given area, like the three wetlands in Skidmore's parcel. WESP generates two scores depicting the ecological condition and risk for each wetland assessed. Ecological condition describes how well the wetland provides its ecosystem services, and risk describes if that wetland is at risk of having the services it provides compromised. The scores generated are out of 10, and can be used to compare the wetlands within an area to help inform conservation strategies.

There are many different methods for rapidly assessing ecosystem services of wetlands within larger areas. We chose to use WESP as our rapid assessment method for evaluating the ecosystem services of the wetlands within the parcel because it met four criteria outlined in a 2007 study of 40 different wetland rapid assessment methods. According to the study, a rapid assessment method for wetlands must be able to measure ecological condition, be "truly rapid", be verifiable, and include a site visit (Fennessy et al. 2007). WESP meets all of these criteria, and is verified by its implementation with government organizations in Canada. WESP has been used in Nova Scotia and Alberta to evaluate wetlands over large areas of natural land. WESP's easily accessible model involves using land cover data as well as site visits to generate scores that depict a wetlands' ecological condition and risk.

Evaluating wetlands using WESP involves completing three forms; Form OF (Office), Form F (Field), and Stressor (S). Each form requires different styles of data collection, most of which can be obtained through GIS and/or Google Earth, except for Form F which requires in-the-field assessment. WESP uses science-based logic models that output scores for the wetlands' abilities to support water storage, streamflow support, water cooling, sediment retention and stabilization, phosphorus retention, nitrate removal and retention, carbon sequestration, organic nutrient export, and habitat for aquatic invertebrate, anadromous fish, non-anadromous fish, amphibians, reptiles, waterbirds, songbirds, raptors, mammals, pollinators, and native plants (Adamus, 2016). The large parcel size and varied wetland types allowed the output of the WESP model to be useful and informative in our study.

Carbon Sequestration

Due to the time limitations and the approach of our study, we only measured carbon storage at a single point in time. To estimate the sequestration values, we used national United States Forest Service data of stands' carbon stocks as they age after a clearcutting event (Smith et al., 2006). Using our preliminary findings, we estimated the stand to be approximately 45 years old. Using the different Northeastern forest types, we estimated the values for each of our C-CAP land cover types currently, and then after 10, 30, 50, and 70 years (Figure C2). We used InVEST software, an open source modeling suite developed by The Natural Capital Project to develop a spatially explicit scenario to estimate carbon storage (Version 3.2.2). Using the InVEST carbon storage model we input the most current C-CAP (2010) land cover data and assumed that the land cover type wouldn't change overtime, but the carbon stored in the forest would. We priced the carbon at 3$ and a 3% discount rate to match the RGGI numbers from the most recent market (Luckow et al., 2016; House et al., 2013). Since its inception in 2009 RGGI has sold 14,791,315 credits to Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New York, Rhode Island, and Vermont. The non-profit cooperative is an emission and allowance tracking system that records CO_2 emissions and CO_2 market allowance transactions. RGGI is the first market-based program in the United States to require the reduction of greenhouse gas emissions (RGGI, 2017). RGGI institutes a CO_2 emissions budget that caps the allowable expulsion of CO_2. Under the cap fossil fuel based electric power generators of 25 megawatts or greater must obtain CO_2 emission allowances for a three-year period. RGGI holds quarterly regional CO_2 auctions selling carbon credits. The proceeds of the carbon credits support energy efficiency and the expansion of renewable energy. RGGI allows companies to use offsets of greenhouse gas emissions reduction and carbon sequestration projects outside the energy sector to meet their carbon quotas. Thus, we measured the market value of the parcel's carbon sequestration regulating service.

General Ecosystem Service Valuation

Due to our time constraint and limited resources, we used land cover based estimates of ecosystem services to gauge a broader estimate of the ecosystem service value of this land. The Trust for Public Land's report on New Hampshire's return on their investments in land conservation provides us with such estimates (Trust for Public Land, 2014). While this data is based in New Hampshire, their similar forest types and regional climates give us reasonable estimates of the land's value in ecosystem services. Using their annual per acre value estimates for deciduous forest, mixed forest, evergreen forest, shrub/scrub, woody wetland, emergent herbaceous wetland, and open water land covers, we multiplied these values by their acreage within Skidmore's parcel to get a total value estimate for our parcel.

Limitations

There is no clear recognition of the value of ecosystem services at the local scale (Hein, 2014; Pandeya et al., 2015). Thus, the results of this case study are mainly illustrative and an experimental rendition of the economic value of ecosystem services. Additionally, since the land parcel is undeveloped, some ecosystem services may be beyond the scope of this case study. Sequestration values are calculated from national estimates and are for after a clear cut has occurred. Additionally, we estimated the forest age, and assumed that there were no factors that were limiting forest growth or carbon storage. For instance, the soils in our plot are relatively shallow, which could limit growth in different areas. Our sequestration values ignore any sequestration by the wetlands. We used rapid assessment methods because of time and resource constraints, which prevented a more thorough extended quantification of ecosystem services. With more time and resources, ecosystem services should be measured over an extended period of time to collect data regarding dynamics and changes within the ecosystem. Our study also omitted measuring areas of ecosystem service value such as wildlife habitat, pollination, nutrient cycling, etc. While the survey did encompass all four categories of ecosystem services, the survey did not elucidate all the ecological processes found in the parcel.

Results

WESP Values

Since the wetlands within the parcel are all unique (sedge, woody, open-water), WESP generated different scores for each. The sedge wetland scored the highest for ecological condition and the lowest for risk (8.15, 3.33), the woody wetland had the lowest value for ecological condition and the mid-value for risk (5.76, 3.83), and the open water wetland had the mid value for ecological condition and highest value for risk (7.57, 4.48; Table 4).

Wetland Type	Ecological Condition	Risk
Sedge	8.15	3.33
Woody	5.76	3.83
Open Water	7.57	4.48

Table 4. *WESP values for ecological condition and risk for the three distinct wetlands within the parcel. Scores are all out of 10. Ecological condition reflects the wetlands abilities to provide its ecosystem services, and risk reflects how easily the wetlands' services it provides can be compromised.*

Carbon Storage

We measured 4,147 trees across our 30 plots, ranging from 33 to 233 trees per plot (Figure C1). Trees ranged from 2.5 to 60.9 cm in diameter. Across the three types of forests, the mixed forest had the highest carbon storage per hectare (340 Mg C ha-1) followed by evergreen forests (324 Mg Cha-1), and deciduous forests (283 Mg C ha-1) (Table 5). All three forests had the most storage within the soils (51 - 64%) followed by the aboveground tree biomass (28 - 38%), the belowground roots (7 - 9%), and nearly equally small amounts in the leaf litter and the forest floor (0.5 - 1.3%) (Figure 2). Most of the soil carbon was within the top 10 cm layer in the deciduous, mixed, and evergreen forests (34%, 36%, 41% respectively).

Wetlands had over half of the parcel's carbon storage (28,988 Mg C of 62,727 Mg C). The forested wetland, in the center of the parcel, had the highest storage per hectare (1080 Mg C ha-1) and contributed 12,731 Mg C to the parcel's overall carbon storage. The emergent wetland in the north, had a similarly high value of 899 Mg C ha-1, but since it covered a larger hectarage, contributed 16,185 Mg C to the parcel's overall carbon storage. Both of these palustrine wetlands had peat 1.00 to 1.66 m deep, and an estimated average depth of 1.40 m. The open water wetland in the south of the parcel the peat was only 40 cm deep with a lower carbon content. The open water land coverage only contributed 72 Mg C.

	Deciduous Forest	Mixed Forest	Evergreen Forest	Emergent Wetland	Forested Wetland	Open Water	TOTALS:
C Mg per hectare	283	339	324	899	1080	72	2,998
% Total Hectarage	28%	30%	19%	13%	8%	1%	138
Mg Carbon	11,112	14,174	8,454	16,185	12,731	72	62,727
Mg CO2 equivalent	40,781	52,017	31,026	59,398	46,723	262	230,208

Table 5. *Carbon storage across the main six land cover types. The carbon per hectare was averaged across plots and is the sum of each component (soil, aboveground tree, belowground root, etc.) measurement. Using the total hectarage from the land cover rasters the total carbon storage across the parcel was measured. The CO_2 equivalent is x 3.67 the carbon content.*

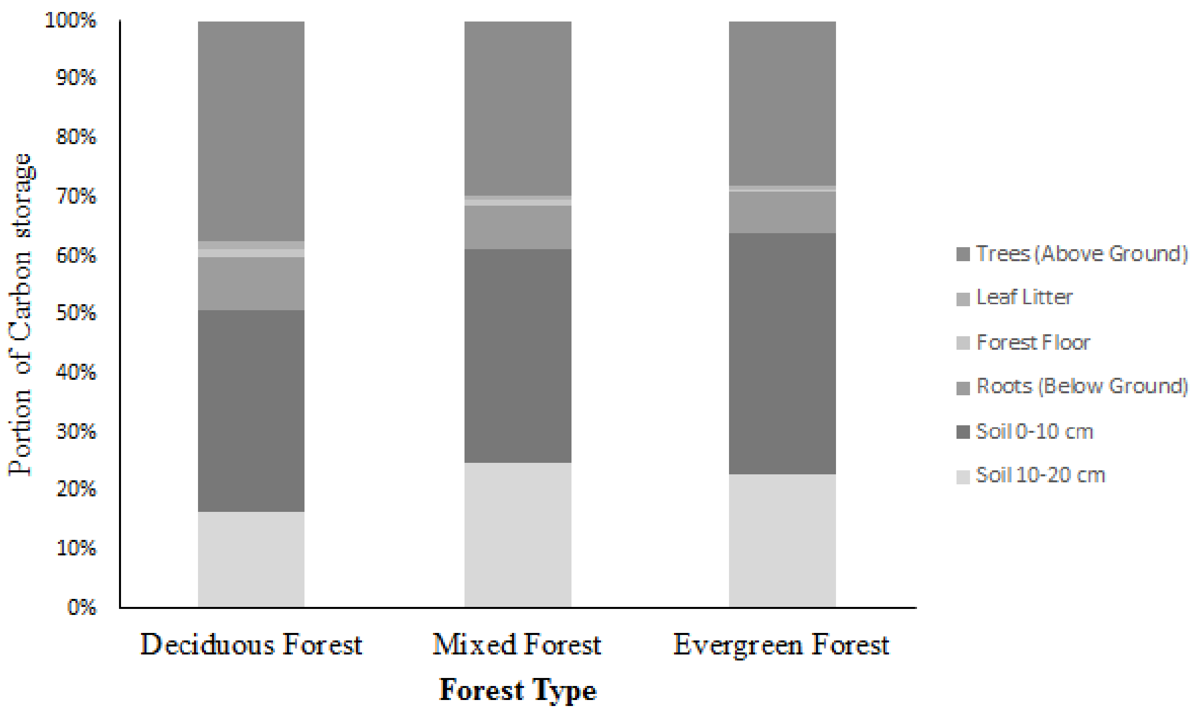

Figure 2. *Carbon stored in each of the forest components measured. Soil stores the most carbon, with evergreen forests storing a higher percentage in their soils than deciduous or mixed forests.*

Carbon Sequestration

Looking at carbon storage over time, or carbon sequestration, we estimate our parcel can store 8,924 Mg of carbon over 70 years or 127 Mg of carbon per year on average (Figure 3a). Sequestration rate declines slightly over time. Deciduous forests sequestered more carbon (9.2

Mg C ha^{-1}) than Mixed forests (8.0 Mg C ha^{-1}) and Evergreen forests (4.6 C Mg ha^{-1}). At a 3$ price, and a 3% discount rate the parcel is worth $11,472 over 70 years (Figure 3b). However, the value of the parcel declines overtime from $436 annually in its first 10 years to only $19 annually over the last 20 years.

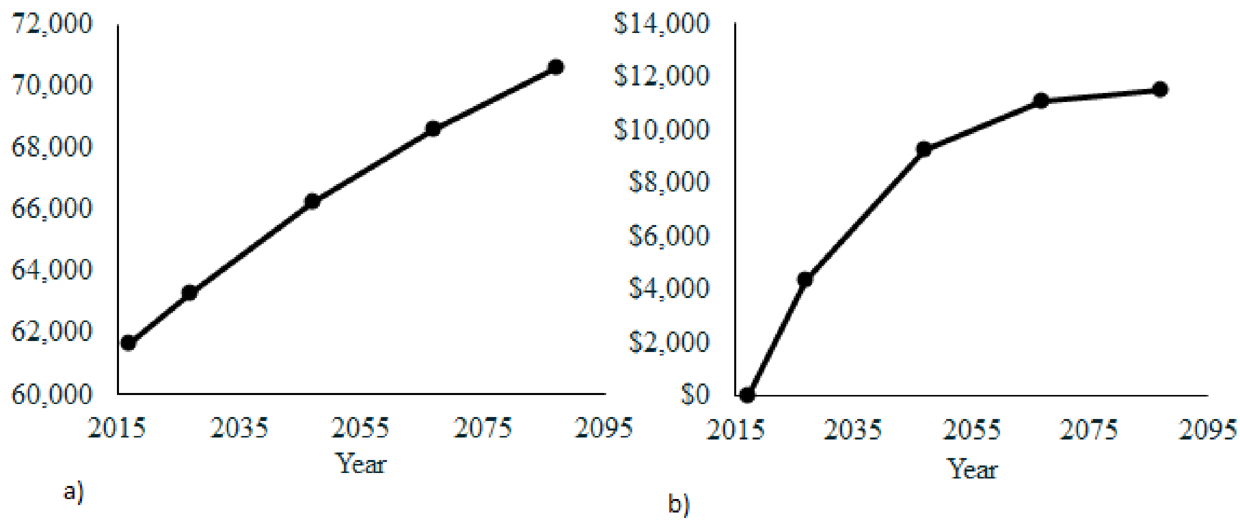

Figure 3. *a) Carbon storage overtime illustrating the sequestration of the forested portion of our parcel at 10, 30, 50, and 70 years. b) The sequestration value of our parcel over time assuming a 3$ price and a 3% discount rate (Luckow et al., 2016; House et al., 2013). Values from Smith et al. (2006) were analyzed in the InVEST software using C-CAPP land cover data.*

Stakeholder Perceptions

Student perception of ecological value was partially assessed by quizzing student's knowledge of different wetland and forest characteristics and functions, in addition to their broader understanding of the term ecosystem services. Overall, students performed well on the quiz. In the wetland section of the quiz, 95% knew that wetlands filter pollutants, 92% knew that wetlands improve drinking water quality, 89% knew that wetlands reduce the cost of severe flooding, 89% knew that wetlands provide wildlife habitat, 57% knew that 34% of all U.S. plant species are found in wetlands, and 48% knew that plants in wetlands absorb excess nutrient inputs. In the forest section of the quiz, 95% knew that forests provide climate regulation, 95% knew that biodiversity loss can impair forest functioning, and 82% knew that trees, leaf litter, and soil store carbon. When asked more generally about the meaning of the term ecosystem services, 94% knew that ecosystems are goods and services that provide local and global benefits, and 91% knew that land management plans affect ecosystem services. One particularly striking finding in the survey, is that despite excellent performance on the quiz, 48% of students responded they did not understand the relationship between ecosystem health and ecosystem services.

When asked if national measures of wealth should include ecosystem services in the gross domestic product as an economic sector, 43% of students agreed and 16% strongly agreed, 20% did not know, 19% disagreed, and 1% strongly disagreed. When asked if dollar estimates strengthen an environmental conservation agenda, 11% strongly agreed, 41% agreed, 30% did not know, 17% disagreed, and 1% strongly disagreed. Students were also asked what they thought was the most important ecosystem service, 46% choose regulating services, 32% marked provisioning services, 19% selected supporting services, and cultural services was ranked last with just 3% of the vote.

Stakeholder Valuations

We used three metrics in order to demonstrate the three broad methods ecological economists employ to value the environment. When students were asked how much of their tuition they would like to allocate towards ecosystem services in the aforementioned parcel, students on average reported $3,219 (Table 6).

Methodology	Metric	Value
Market Based	Established market: RGGI carbon credits	$164
Revealed Preference	Annual mountain biking expenditure: SMBA	$676
Stated Preference	Skidmore students willingness to pay for ecosystem services allocated from Skidmore College tuition ($66,450)	$3,219

Table 6. Ecosystem service stakeholder valuations according to methodology

When students were asked which of the above metrics is the most suitable measure of ecological value, 48% indicated market based as the best method, 30% said none of these methods are suitable measures of ecological value, 27% voted for stated preference, and 19% chose stated preference. Even though 30% of students thought none of the methods are suitable measures of ecological value, 73% of students thought at least some of the time economic values should represent the services ecosystems provide to humans.

General Ecosystem Service Valuation

Using The Trust for Public Land's (2014) report on New Hampshire public lands, we estimate the parcel's total value of ecosystem services to be worth $150,896 annually. Woody and Emergent herbaceous wetlands were valued at $1,289/acre each while the three forest types and open water were valued at $205-$234/acre (Table 7). Wetlands were worth nearly 6 times as much as forests and even though wetlands make up a small portion of the parcel's land, their high value makes up almost ⅔ of the parcel's total value.

Land Cover	Main Goods and Services	Value per acre	Parcel Acreage	Value
Deciduous forest	Air pollution removal and carbon sequestration	$205	97	$19,878
Mixed forest	Air pollution removal and carbon sequestration	$210	103	$21,670
Evergreen forest	Air pollution removal and carbon sequestration	$215	64	$13,866
Shrub/scrub	Air pollution removal and carbon sequestration	$18	1	$20
Woody wetland	Water quality and wildlife habitat	$1,289	29	$37,553
Emergent herbaceous wetland	Water quality and wildlife habitat	$1,289	44	$57,333
Open water	Freshwater regulation and supply and wildlife habitat	$235	2	$575

Table 7. *Land cover types their ecosystem services and ecosystem service values. The total parcel value is $150,896. From The Trust For Public Land (2014).*

Discussion

WESP

Currently, the open-water wetland and its risk rating of 4.48/10 indicates that this wetland is susceptible to degradation when land around it is not conserved. The area all along the parcel's eastern boundary has been recently logged right up to the sedge wetlands' eastern edge. If logging, or a lack of directed management continues the services the wetlands provide could be compromised. These wetlands are important not only to the health of nearby ecosystems, but also to Saratoga Springs' drinking water source, Loughberry Lake. The wetlands are adjacent to Loughberry Lake's watershed boundary. Interestingly, an active beaver dam is directing the water of the open-water wetland away from the Loughberry Lake watershed and instead into the Kayaderosseras Creek Watershed, demonstrating the need for further monitoring, study, and management of this changing ecosystem. Scores from WESP can help inform landowners about ecological health of wetlands. If future development impacts wetlands' ecological health, its services may be at risk. The wetlands within the parcel should be continuously monitored using WESP, and the scores generated should influence how Skidmore manages the land within the parcel.

We recommend that Skidmore acknowledge WESP as a valid method for assessing its wetlands in all of its parcels owned, and continuously monitors its wetlands using WESP. The wetlands within the Skidmore parcel are also adjacent to Department of Environmental Conservation (DEC) owned wetlands to the west. Skidmore should work with DEC and assess the wetlands using WESP over the greater area around the parcel. WESP is easily used and accessible to people who are not wetland experts, and, can be used as a tool for monitoring wetlands by almost anyone.

With the active changes in the wetlands' ecosystems, and the current risk rating of the open water wetland, continued monitoring of the parcel's wetlands is essential towards ensuring the safety of its ecosystem services. By embracing this model for rapidly assessing wetlands, Skidmore can set a precedent of managing wetlands for something beyond their mere existence, and focus on the status of the ecological services the wetlands provide. WESP allows decisions to be made on quantifiable measures of the wetlands' ecological health in the context of its surrounding areas. Having Skidmore adopt this strategy would be a step in shifting the emphasis of ecosystem services from broad national and global calculations to localized assessment and inventories that will help land managers on-the-ground make scientifically supported decisions.

Carbon Storage

Carbon storage estimates set a baseline for determining the carbon budget of a forest that helps future measurements for carbon fluxes or energy transfers within an ecosystem. While the trees are significant portions of carbon in our plot, they are only about one third of the carbon storage. Soils are the primary carbon storage component holding nearly 50% of the ecosystems' carbon. This allocation of carbon budgets reflects the pattern across most U.S. forests (Turner et al., 1995). The forest floor and leaf litter made up surprisingly small components compared to literature estimates of 6-10% (Birdsey, 1992; Turner et al., 1995). This could be evidence of this land's logging history or a disturbance in the ecosystem functioning. The leaf litter decomposing into the forest floor is eventually incorporated into deepening organic layers within the soils, where carbon storage can be stored for extended periods. Annual leaf litter turnover is critical to adding carbon and soil health and nutrients overall. Harvesting tree stocks can expose soils and cause the soils to decline in carbon content, but these effects can be mitigated (Nave et al., 2010).

We saw some of the effects of logging first hand at plot number 12, which was within the logged section of parcel. This plot lacked a forest floor layer and significant carbon storage in tree biomass, resulting in the lowest carbon storage of all of the plots. However, the soils within the plot had similar carbon content to other plots, but lack any organic layer and were very dry. Since the logging event occurred about one year ago, the full effects of the logging may not be seen in the soil immediately.

Carbon storage was much higher in wetland peats as wetlands represent major carbon sinks across the world's' landscapes. Our values (899 - 1080 Mg C ha^{-1}) were slightly lower than the countrywide average carbon density of 1,500 Mg C ha^{-1}, but still lie within the expected range (Bridgham et al., 2006). The forested and emergent herbaceous wetlands were of similar depths of 1.00 to 1.22 m near the edges and 1.4 to 1.66 m towards the centers. These depths are similar to those of other Adirondack region wetlands suggesting some controlling factor limiting the peat depth, and thus the carbon storage in the region (Kurt Smemo, personal communication, 2017). Any conclusions on the carbon budget or contribution to carbon sequestration the wetlands have was outside the purview of this study and include diving into what the range of factors that control the internal carbon cycling are (Zhang et al. 2002).

The wetlands in our study site are three distinct types, representing different capacities for carbon storage. How these wetlands have developed and will change is critical to understanding the parcel's carbon storage and sequestration potential. The open water wetland is especially interesting with its beaver dam on its eastern edge which is controlling its flow dynamics and water levels. Given the variety of types of wetlands within a close proximity to each other, these wetlands have immense potential for future studies. Deeper analysis of their total ecosystem service value, and impact on the Loughberry Lake watershed are two areas of study. Additionally, a future study could examine the precise carbon budgets of these three different types of wetlands, which could provide a distinctive arena to tease out driving factors among the complex set of climatic and biogeochemical variables that have shaped each wetlands' ecological characteristics.

Carbon Sequestration

Sequestration of the forest was measured using national forest carbon stock estimates and applied to the land cover of our parcel. Given the forests' sequestration ability, this parcel presents an opportunity for a possible offset of Skidmore's GHG emissions. Comparing this average yearly rate to Skidmore's emissions, the forested parcel can provide an estimated 2.8% offset of Skidmore's 16,972 Mg of CO_2 equivalent emissions (Figure 4; Skidmore Sustainability Office, 2013). The sequestration of this parcel is equivalent to removing an average of 100 cars off the road (U.S. EPA, 2014). This is a smaller estimate compared to a 2012 Capstone which only looked at 81 of the total 138 hectares (Freeman et al., 2012). Their estimate was based on field measurements of Primary Ecosystem Productivity which is likely more accurate and site-specific than estimates from nationwide data. Our study's approach of land cover and USFS data ignores management and environmental changes. This reinforces the underestimates land use change models have globally compared to atmospheric inversion models or site inventories (House et al. 2013).

Given, Skidmore's goal to reduce their GHG gas emissions by 2025, further studies should update the in situ carbon sequestration values for use as an offset in the college's GHG inventory. By understanding forest dynamics, land can be managed to store carbon (Barford et al., 2001). In order to manage a forest to maximize its offset potential, a clearer understanding of the fluxes in these carbon storages is needed. The changes in carbon storage overtime are poorly understood, but stands to serve as a critical sink of carbon (Goodale, 2002). Debates over the best management practices on maximizing sequestration vary greatly, specifically with the challenge of forests' slow growth rate and complexity of ecosystem interactions. Monitoring this parcel's carbon storage over time is an ideal first step to determining the carbon dynamics of this forest.

There is a clear need for quality data when using any type of model. Without quality data, studying ecosystem services and their values becomes fraught with misleading assumptions. Only about one third of ecosystem service studies have the proper biophysical assessments to draw sound conclusions (Seppelt et al., 2011).

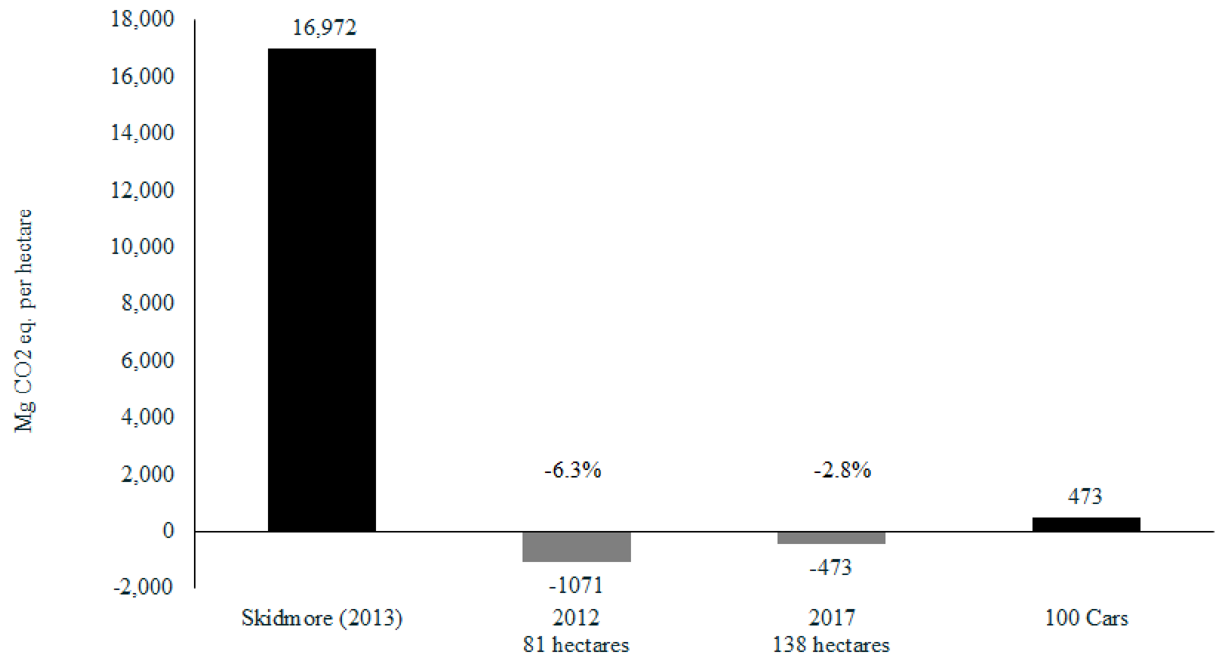

Figure 4. *Comparing Skidmore's Greenhouse Gas emissions to two different sequestration offsets. The 2012 estimate was a portion of the total parcel, but used in situ methods. The 2017 estimate was the full parcel size, but used national scale carbon estimates from the USFS. This land, if conserved, would offset the equivalent of 100 cars annually.*

Stakeholder Perceptions

The impact the health of an ecosystem has on its functioning and ability to provide goods and services is one of the central questions raised in the literature on ecological economics. The fact that most students understood the necessary ecological processes, but did not understand the relationship between ecosystem health and services corroborates the issue raised in the literature by Chaudhary et al. (2015) and Vira & Adams (2009). Another claim made by the literature that is not supported by our research is that people value cultural services more easily than regulating and supporting services. The literature (MA, 2005) argues that people more readily understand the economic value of provisioning and cultural services because those benefits are less abstract than regulating and supporting services. This finding is a result from our sample population and is not meant to be interpreted as a conclusive statement. However, this finding does reveal that our sample population is more biocentric and altruistic than the people surveyed in the broader literature. Prioritizing regulating services as the most important ecosystem services indicates that Skidmore students care about ecological processes impacting the climate.

Another unique finding of our study is that most students thought national measures of wealth should incorporate ecosystem services into the gross domestic product. This is a promising finding that aligns with the UN recommendation to include ecosystem services into national accounts. This raises the question, how can Skidmore incorporate ecosystem services into their college account?

The most contentious question in our survey, and likely the most heated issue in the discipline of ecological economics is if dollar estimates of ecosystem services strengthen the environmental conservation agenda. In our survey, 51% of students either agreed or strongly agreed with dollar estimates, while very few students disagreed (17%) or strongly disagreed (1%). However, it is notable to mention 30% of students did not know if economic values strengthen the environmental conservation agenda. One finding consistent across the survey is the fact that one third of all students did not know the answers to the questions meant to gauge student perception of ecological value. This finding is representative of the greater disconnect between the benefits the environment provides to society and the ways society excludes the environment from market systems.

Stakeholder Valuations

The dollar estimates we provide are not meant to serve as absolute measures of ecological value. Rather, the estimates we obtain from market based, revealed preference, and stated preference are meant to function as representations of the ways people benefit from ecosystem services. By including a market based estimate of carbon sequestration we convey the monetary gain from regulating services. Likewise, by showing the way SMBA mountain bikers spend money mountain biking, we showcase the value of cultural services. Lastly, the stated preference estimate assesses the way students value all four ecosystem services, by asking students more generally about how much money Skidmore College should devote to ecosystem services. The stated preference estimate is the largest because students were asked to consider the cost of Skidmore College tuition ($66,450) in context to spending on ecosystem services. If students were asked how much they would be willing to pay out of pocket, students would likely respond with a much smaller amount. This would occur because the average student's disposable income is much smaller than the annual cost of tuition. Our estimate from revealed preference could be explained by the sample size surveyed. With a larger sample size of people who live further away from the Daniels Road State Forest trailhead, but also use the SMBA trails, the value could change. Lastly, our market based estimate would be much larger if it included all of the forested area owned by Skidmore College. This is worth noting, because Skidmore currently lacks an accurate estimate of its total offset capacity, and our estimate is only indicative of the studied parcel's carbon sequestration. Overall, these three estimates of ecological value are meant to make the benefits ecosystems provide to humans more visible to the greater Skidmore community.

General Ecosystem Service Valuation

Using the inclusive estimates done in the nearby New Hampshire region, we estimated the overall value of our parcel's ecosystem services to be $150,896 annually. This is much larger than any of our other methods. Seeing the value of forests and especially wetlands, will hopefully help Skidmore continue to accept future undeveloped land in the future. Continuing this study to include a full array of ecosystem service assessments and their value will help direct the college in protecting and valuing their land with due diligence in the future. Completing such a comprehensive study, like The Trust for Public Land did in New Hampshire, is critical to helping local and regional land managers and the communities they serve understand the value of the lands they are choosing to protect.

Skidmore's Land Management and Regional Issues

In the past 2 years, development has begun to encroach upon the forest surrounding Skidmore's land parcel. Hilltop Drive, an access road, was constructed in 2015. The adjacent parcel along the eastern border of the Skidmore parcel was logged in 2016, and continuing with logging of the adjacent parcel in 2016, leaving disturbed ecosystems all along the eastern border of the parcel (Appendix B). Hilltop Drive has made both Skidmore's parcel and the nearby parcel which may soon have large estate houses more accessible. With Skidmore's access rights, Hilltop Drive could be a good location to establish a trailhead and kiosk.

Because of this recent logging and potential future development, the parcel's ecosystems will change. Storm water runoff will increase, wildlife habitats and behaviors will shift, and the parcel could become susceptible to invasive species establishment. Skidmore is facing a pivotal time for land conservation.

By conserving this land parcel, Skidmore can claim offsets to its carbon emissions through naming the land as a carbon sink. As mentioned before, there are debates about best practices for managing forests as carbon sinks. However, Skidmore should conserve this land for its carbon dioxide sequestration potential, and utilize its offsets to help lower the college's GHG emissions and meet its Energy goals outlined in its Campus Sustainability Plan for 2025.

In the past, Skidmore forested lands have been conserved for research and educational opportunities (Freeman et al. 2012). In accomplishing Focus Area 5 of the Campus Sustainability Plan, Skidmore must continue to engage students, faculty, employees, and community members with the college's sustainability initiatives (Campus Sustainability Subcommittee, 2015). Skidmore should promote the use of this land for research and recreation. Some actions that Skidmore can take to encourage awareness and use of its land related sustainability efforts are hosting volunteer community events to survey, explore, and measure some ecosystem services of the parcel. Through these events, people can engage with a biologically diverse area and study the quantification of ecosystem services further.

Focus Area 4 states that Skidmore will have comprehensive land management plans for all of its owned parcels by 2025. In the past, Skidmore has allocated resources towards conservation of its Northwoods through establishing a Northwoods Steward position. If Skidmore were to grant oversight of land conservation as well as allocate more resources to the Skidmore Sustainability Office, another land steward can be hired to maintain conservation goals for the new parcel. In particular, the parcel we studied provides the opportunity for Skidmore to develop a framework of conservation based on ecological assessment and ecosystem service values. We recommend that Skidmore conserve this parcel with its ecosystem services as a baseline for future conservation practices. Conserving land because of its ecosystem services sets a precedent for other institutions similar to Skidmore. By conserving the parcel for its ecosystem services, Skidmore can pave the way for other local landowners and small institutions to accept ecosystem services within their market values.

This parcel is located within the larger Palmertown Conservation Area, which extends from SPA State Park north to Moreau Lake State Park (Figure 5). Since, Saratoga PLAN identified this 40,000 acre area of intact forest, SUNY Albany students have begun a preliminary study of the area. The Palmertown Trail, proposed within this area, hopes to provide a thru-hiking or biking trail that could connect to portions of the state government funded Empire State Trail, Hudson River Valley Greenway Trail, Champlain Canalway Trail, and Erie Canalway Trail (Saratoga PLAN, 2017). The vision of a regional Conservation Area and thru-trails is still in initial planning phases and lacks an organized group to drive plans moving forward.

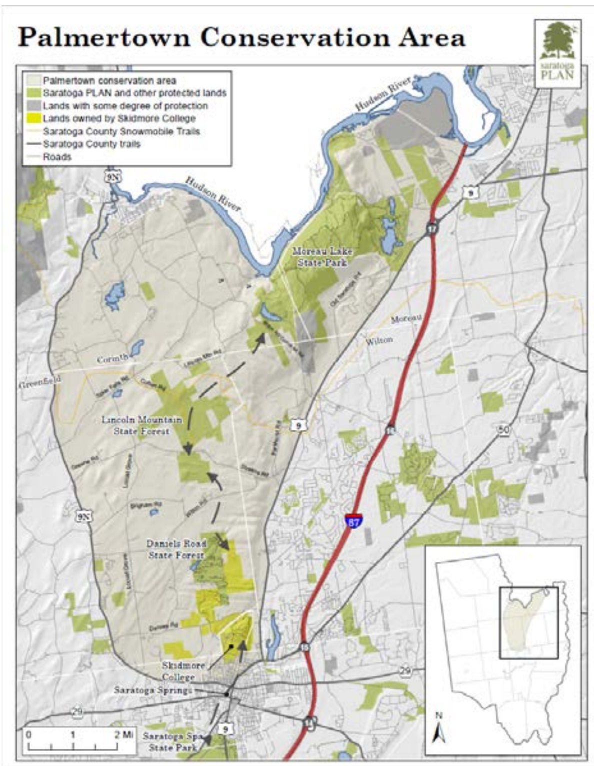

Figure 5. Palmertown Conservation Area map created by Saratoga PLAN (2017). Our study parcel is highlighted east of Daniels Road State Forest.

The Skidmore parcel lies at the gateway into the Palmertown Conservation Area from Saratoga Springs. Given this location, Skidmore's land management plan and decisions moving forward will have a prominent and active role in determining the outcome of the larger area's future conservation and management. If Skidmore were to adapt ecosystem services as a key aspect in their land management plan framework, these guiding principles would ripple into any future conservation plans within the region. Skidmore has an opportunity to use its students and resources to cooperate with community organizations and landowners to drive forward an innovative and comprehensive conservation and recreation vision.

Conclusions

While it is uncertain how climate change will affect natural land in the Capital Region, it's clear that development will drastically alter natural ecosystems. Skidmore's undeveloped land parcels are situated in areas that are ripe with progressive development. Whether it be logging, or construction of luxury condominiums, Skidmore College faces a pivotal moment regarding its land management strategies of its parcel within the Palmertown Conservation Area. Skidmore, in the midst of developing their land management plan, has the opportunity to take valuable strides in conservation. At a minimum, monitoring and inventorying Skidmore's natural resources should be the first step to managing and protecting the ecosystems present in this parcel. This would help the college understand the resources they already own, and also address how best to protect this land in the future. We urge Skidmore College to develop a land management plan with an emphasis on strong scientific assessments of ecosystem services and their values. Given Skidmore's role as a higher-learning institution, we suggest the college go further, and utilize our methods for evaluating and monitoring Skidmore's ecosystem services.

References

Adamus, P.R. (2016). Manual for the Wetland Ecosystem Services Protocol (WESP). people.oregonstate.edu/~adamusp.

Alexander, J. G., Jirón, D. J., & Christman, C. (2014). United States Department of Agriculture. Land Management Plan 2014 Revision: Shoshone National Forest. Retrieved from http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5446327.pdf.

Alley, R. B., Marotzke, J., Nordhaus, W. D., Overpeck, J. T., Peteet, D. M., Pielke, R. A., ... & Wallace, J. M. (2003). Abrupt climate change. Science, 299(5615), 2005-2010.

Barford, C. C., Wofsy, S. C., Goulden, M. L., Munger, J. W., Pyle, E. H., Urbanski, S. P., … Moore, K. (2001). Factors Controlling Long- and Short-Term Sequestration of Atmospheric CO2 in a Mid-latitude Forest. *Science*, 294(5547), 1688–1691.

Bateman, I. J., Carson, R. T., Day, B., Hanemann, M., Hanley, N., Hett, T., & Pearce, D. W. (2002). Economic valuation with stated preference techniques: a manual.

Birdsey, R. A. (1992). Carbon storage and accumulation in United States forest ecosystems. *Gen. Tech. Rep.* WO-59.

Bridgham, S. D., Megonigal, J. P., Keller, J. K., Bliss, N. B., & Trettin, C. (2006). The Carbon Balance of North American Wetlands. *Wetlands*.

Brown, G., & Mendelsohn, R. (1984). The hedonic travel cost method. *The review of economics and statistics*, 427-433.

Brown, S., Schroeder, P., & Kern, J. (1999). Spatial distribution of biomass in forests of the eastern USA. *Forest Ecology and Management*, 123 (1), 81–90.

Cai, Y., Judd, K. L., Lenton, T. M., Lontzek, T. S., & Narita, D. (2015). Environmental tipping points significantly affect the cost− benefit assessment of climate policies. *Proceedings of* the National *Academy of Sciences*, 201503890.

Cairns, M. A., Brown, S., Helmer, E. H., & Baumgardner, G. A. (1997). Root biomass allocation in the world's upland forests. *Oecologia*, 111(1), 1–11

Chaudhary, S., McGregor, A., Houston, D., & Chettri, N. (2015). The evolution of ecosystem services: A time series and discourse-centered analysis. *Environmental Science & Policy*, *54*, 25-34.

Costanza, R., d'Arge, R., De Groot, R., Faber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'neill, R.V., Paruelo, J. & Raskin, R. G. (1997). The value of the world's ecosystem services and natural capital.

Costanza, R., Stern, D., Fisher, B., He, L., & Ma, C. (2004). Influential publications in ecological economics: a citation analysis. *Ecological Economics*, *50*(3), 261-292.

Costanza, R., De Groot, R., Sutton, P., van der Ploeg, S., Anderson, S. J., Kubiszewski, I., Farber, S., & Turner, R. K. (2014). Changes in the global value of ecosystem services. *Global Environmental Change*, *26*, 152-158.

Creswell, J. W. (2012). Qualitative inquiry and research design: Choosing among five approaches. *Sage publications.*

De Groot, R., Brander, L., Van Der Ploeg, S., Costanza, R., Bernard, F., Braat, L., Christie, M., Crossman, N., Ghermandi, A., Hein, L. and Hussain, S. (2012). Global estimates of the value of ecosystems and their services in monetary units. *Ecosystem services*, *1*(1), pp.50-61.

Daily, G. (1997). Nature's services: societal dependence on natural ecosystems. *Island Press.* Washington D.C.

Daly, H. E. (1992). Allocation, distribution, and scale: towards an economics that is efficient, just, and sustainable. *Ecological economics*, *6*(3), 185-193.

EPA. (2014). Greenhouse Gas Emissions from a Typical Passenger Vehicle. Office of Transportation and Air Quality. Green Vehicle Guide. Retrieved from https://www.epa.gov/sites/production/files/2016-02/documents/420f14040a.pdf.

Fankhauser, S., Tol, R. S., & Pearce, D. W. (1997). The aggregation of climate change damages: a welfare theoretic approach. Environmental and Resource Economics, 10(3), 249-266.

Farber, S. C., Costanza, R., & Wilson, M. A. (2002). Economic and ecological concepts for valuing ecosystem services. *Ecological economics*, *41*(3), 375-392.

Farmer, J., Matthews, R., Smith, P., Langan, C., Hergoualc'h, K., Verchot, L., & Smith, J. U. (2014). Comparison of methods for quantifying soil carbon in tropical peats. *Geoderma*, 214, 177-183.

Fennessy, M. S., Jacobs, A. D., Kentula, M. E. (2007). “An evaluation of rapid methods for assessing the ecological condition of wetlands”. *Wetlands*. 27 (3). 543-560.

Freeman, R., Slocombe, J., Fillion, E. (2012). “A Gift that Keeps on Giving: An Investigation of the Uses, Functions, and Services of Skidmore College’s New Land”. ES375: Case Studies in Sustainability.

Gómez-Baggethun, E., De Groot, R., Lomas, P. L., & Montes, C. (2010). The history of ecosystem services in economic theory and practice: from early notions to markets and payment schemes. *Ecological economics*, *69*(6), 1209-1218.

Goodale, C. L., Apps, M. J., Birdsey, R. A., Field, C. B., Heath, L. S., Houghton, R. A., … Shvidenko, A. Z. (2002). Forest Carbon Sinks in the Northern Hemisphere. Ecological Applications, 12(123), 891–899.

Hein, L. (2014). Biophysical modeling and and analysis of ecosystem services in an ecosystem accounting context. United Nations Environment Program

Hibbard, P. J., Okie, A. M., Tierney, S. F., & Darling, P. G. (2015). The economic impacts of the Regional Greenhouse Gas Initiative on nine Northeast and Mid-Atlantic states. Analysis Group: Boston, MA.

House, J. I., Prentice, I. C., Ramankutty, N., Houghton, R. A., & Heimann, M. (2003). Reconciling apparent inconsistencies in estimates of terrestrial CO2 sources and sinks. Tellus, Series B: Chemical and Physical Meteorology, 55(2), 345–363.

Hughes, R. F., Kauffman, J. B., & Jaramillo, V. J. (1999). Biomass, carbon, and nutrient dynamics of secondary forests in a humid tropical region of Mexico. *Ecology*, 80(6), 1892-1907.

Jenkins, J. C., Chojnacky, D. C., Heath, L. S., & Birdsey, R. A. (2003). National-scale biomass estimators for United States tree species. *Forest Science*, 49(1), 12–35.

Konen, M. E., Jacobs, P. M., Burras, C. L., Talaga, B. J., & Mason, J. A. (2002). Equations for predicting soil organic carbon using loss-on-ignition for north central US soils. Soil Science Society of America Journal, 66(6), 1878-1881.

Kosoy, N., & Corbera, E. (2010). Payments for ecosystem services as commodity fetishism. *Ecological economics*, *69*(6), 1228-1236.

Lemoine, D. M., & Traeger, C. P. (2012). Tipping points and ambiguity in the economics of climate change (No. w18230). National Bureau of Economic Research.

Luckow, P., Stanton, E., Fields, S., Ong, W., Biewald, B., Jackson, S., & Fisher, J. (2016). Spring 2016 National Carbon Dioxide Price Forecast.

Lenton, T. M., Held, H., Kriegler, E., Hall, J. W., Lucht, W., Rahmstorf, S., & Schellnhuber, H. J. (2008). Tipping elements in the Earth's climate system. Proceedings of the National Academy of Sciences, 105(6), 1786-1793.

Ma, C., & Stern, D. I. (2006). Environmental and ecological economics: A citation analysis. *Ecological Economics*, *58*(3), 491-506.

Macpherson, A. J., Principe, P. P., & Smith, E. R. (2010). A directional distance function approach to regional environmental–economic assessments. *Ecological Economics*, *69* (10), 1918-1925.

Masood, E., & Garwin, L. (1998). Audacious bid to value the planet whips up a storm. *Nature*, *395*(6701), 430.

Millennium Ecosystem Assessment. (2005). Ecosystems and human well-being. *United Nations Environment Program.* Washington, DC.

Nave, L. E., Vance, E. D., Swanston, C. W., & Curtis, P. S. (2010). Harvest impacts on soil carbon storage in temperate forests. Forest Ecology and Management.

NOAA. (2010). C-CAP Land Cover Atlas. Office for Coastal Management.

Nordhaus, W. D. (1994). *Managing the global commons: the economics of climate change* (Vol. 31). Cambridge, MA: MIT press.

Nordhaus, W. D. (2007). A Review of the" Stern Review on the Economics of Climate Change". *Journal of economic literature*, 686-702.

Pandeya, B., Buytaert, W., Zulkafli, Z., Karpouzoglou, T., Mao, F., & Hannah, D. M. (2016). A comparative analysis of ecosystem services valuation approaches for application at the local scale and in data scarce regions. *Ecosystem Services*.

Peterson, M. J., Hall, D. M., Feldpausch-Parker, A.M., & Peterson, T. R. (2010). Obscuring ecosystem function with application of the ecosystem services concept. *Conservation Biology*, *24*(1), 113-119.

Price, P. (2007). An introductory guide to valuing ecosystem services. *Department for Environment, Food and Rural Affairs.* Nobel House, 17.

RGGI. (2017). Regional greenhouse gas initiative: an initiative of the Northeast and Mid-Atlantic States of the U.S. https://www.rggi.org/

Rial, J. A., Pielke Sr, R. A., Beniston, M., Claussen, M., Canadell, J., Cox, P., ... & Salas, J. D. (2004). Nonlinearities, feedbacks and critical thresholds within the Earth's climate system. *Climatic Change*, 65(1-2), 11-38.

Rind, D., 2008: The consequences of not knowing low and high latitude climate sensitivity. *Bull.* Amer. *Meteorol. Soc.*, 89, 855-864

Røpke, I. (2004). The early history of modern ecological economics. *Ecological economics*, *50*(3), 293-314.

Saratoga PLAN. (2017). Palmertown Conservation Area. Retrieved from http://www.saratogaplan.org/explore/public-preserves-trails/palmertown-conservation-area/

Schneider, S. H. (2004). Abrupt non-linear climate change, irreversibility and surprise. *Global Environmental Change*, 14(3), 245-258.

Schröter, M., Zanden, E. H., Oudenhoven, A. P., Remme, R. P., Serna-Chavez, H. M., Groot, R. S., & Opdam, P. (2014). Ecosystem services as a contested concept: a synthesis of critique and counter-arguments. *Conservation Letters*, *7*(6), 514-523.

Seppelt, R., Dormann, C. F., Eppink, F. V., Lautenbach, S., Schmidt, S. (2011). “A quantitative review of ecosystem service studies: approaches, shortcomings and the road ahead.” *Journal of Applied Ecology.* 48. 630-636.

Skidmore Sustainability Office. (2013). Skidmore College Greenhouse Gas Inventory. Retrieved from https://www.skidmore.edu/sustainability/documents/2013SkimdoreGHGReport.pdf

Skidmore Sustainability Committee. (2015). Skidmore College Campus Sustainability Plan. Retrieved from https://www.skidmore.edu/sustainability/documents/2015CampusSustainabilityPlan.pdf

Soil Survey Staff. Natural Resources Conservation Service, United States Department of Agriculture. Web Soil Survey. Available online at https://websoilsurvey.sc.egov.usda.gov/. Accessed 4/18/2017.

Smith, J. E., Heath, L. S., Skog, K. E., & Birdsey, R. A. (2006). A Model of Forest Floor Carbon Mass for United States Forest Types. *US Forest Service General Technical Report In Press*, 42. Retrieved from https://www.fs.fed.us/ne/newtown_square/publications/research_papers/pdfs/2002/rpne722.pdf

Stern, N. H. (2006). Stern Review: The economics of climate change (Vol. 30). London: HM treasury.

Tol, R. S. (1996). The damage costs of climate change towards a dynamic representation. Ecological *Economics*, *19*(1), 67-90.

Tol, R. S. (2009). The economic effects of climate change. *The Journal of Economic Perspectives*, 29-51.

Toman, M. (1998). Special section: forum on valuation of ecosystem services: why not to calculate the value of the world’s ecosystem services and natural capital. *Ecological economics*, *25*(1), 57-60.

TEEB. (2010). The Economics of Ecosystems and Biodiversity: Mainstreaming the Economics of Nature: A synthesis of the approach, conclusions and recommendations of TEEB. *United Nation Environmental Program.*

Trust for Public Land. (2014). New Hampshire's Return on Investment in Land Conservation. Boston, MA. Retrieved from https://www.tpl.org/sites/default/files/nh-state-roi-report.pdf

Turner, D. P., Koerper, G. J., Harmon, M. E., & Lee, J. J. (1995). A carbon budget for forests of the conterminous United States. *Ecological Applications*, 5(2), 421–436.

University of Central Florida. (2014). Natural Areas Land Management Plan: University of Central Florida, Landscape and Natural Resources. Retrieved from https://www.green.ucf.edu/wp-content/uploads/2014/04/2014-Land-Management-Plan.pdf

U.S. EPA. (2014). Greenhouse Gas Emissions from a Typical Passenger Vehicle, (May), 1–5.
Vatn, A., & Bromley, D. W. (1994). Choices without prices without apologies. *Journal of environmental economics and management*, *26*(2), 129-148.

Vira, B., & Adams, W. M. (2009). Ecosystem services and conservation strategy: beware the silver bullet. *Conservation Letters*, *2*(4), 158-162.

Wind, L., & Hole, W. (2013). Initial chemistry and decay dynamics of deciduous leaf litter from a long-term soil-warming study in the Northeastern US.

Winpenny, J. T. (1991). Values for the environment: a guide to economic appraisal. HMSO Publications Centre.

Zadrozny, A. L., & Brenner, J. C. (2011). Ithaca College Natural Lands Management Plan.

Zander, K. K., Ainsworth, G. B., Meyerhoff, J., & Garnett, S. T. (2014). Threatened bird valuation in Australia. *PloS one*, *9*(6).

Appendices

Appendix A - Parcel Maps with key features

Figure A1. *Skidmore's properties outlined (green), and labels of surrounding areas for reference. Image generated with ArcMap using NYS orthoimagery*.

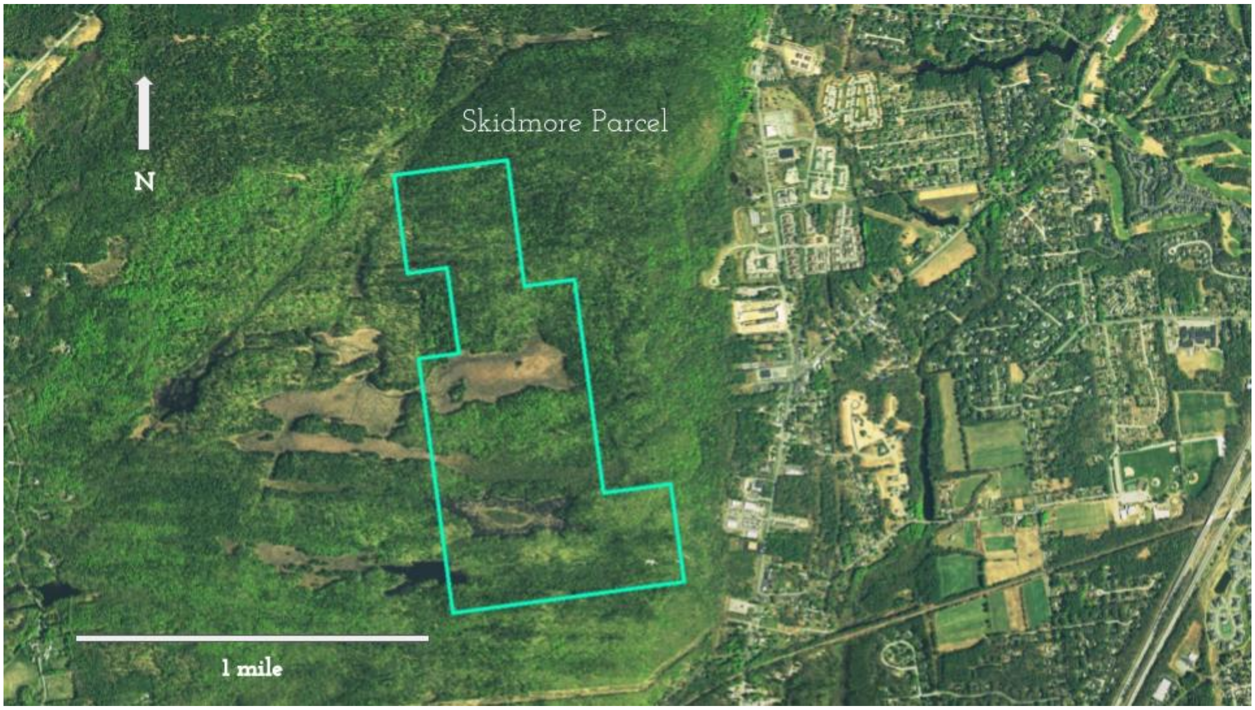

Figure A2. *Skidmore Parcel studied with two of the three wetlands clearly visible. The third, forested wetland, located just north of the open water wetland, is a darker patch of trees that fades into a sedge wetland at the western boundary. Image generated with ArcMap using NYS orthoimagery.*

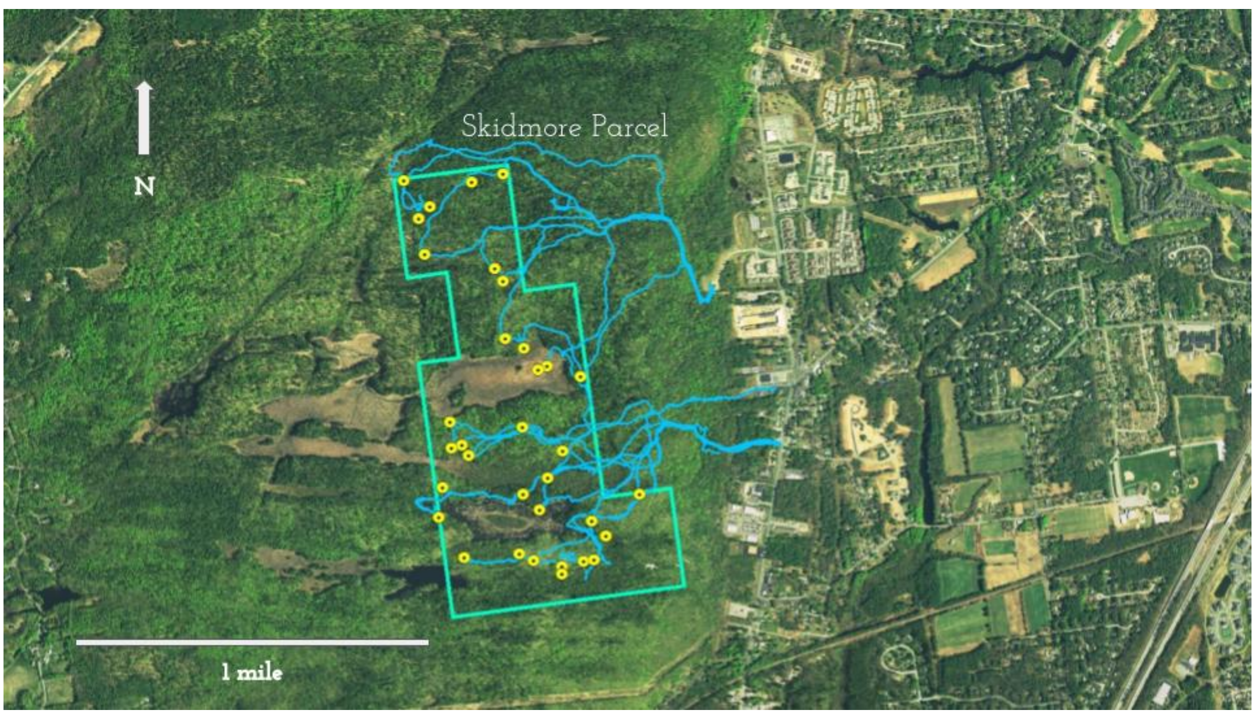

Figure A3. *Parcel, sample points (including 4 wetland survey locations), and routes taken to access sample points. Image generated with ArcMap using NYS orthoimagery.*

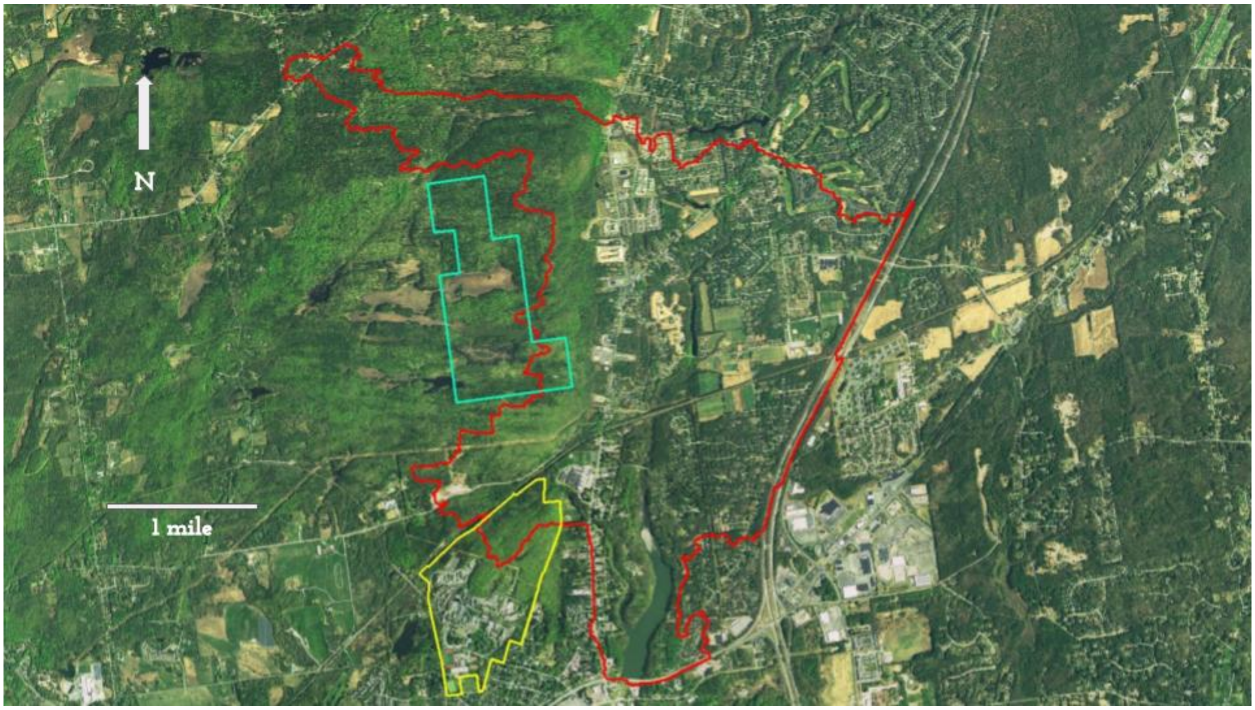

Figure A4. *Satellite image of Loughberry Lake watershed (red), Skidmore College's campus and northwoods (yellow), and Skidmore's new land parcel (blue). Image generated with ArcMap using NYS orthoimagery*.

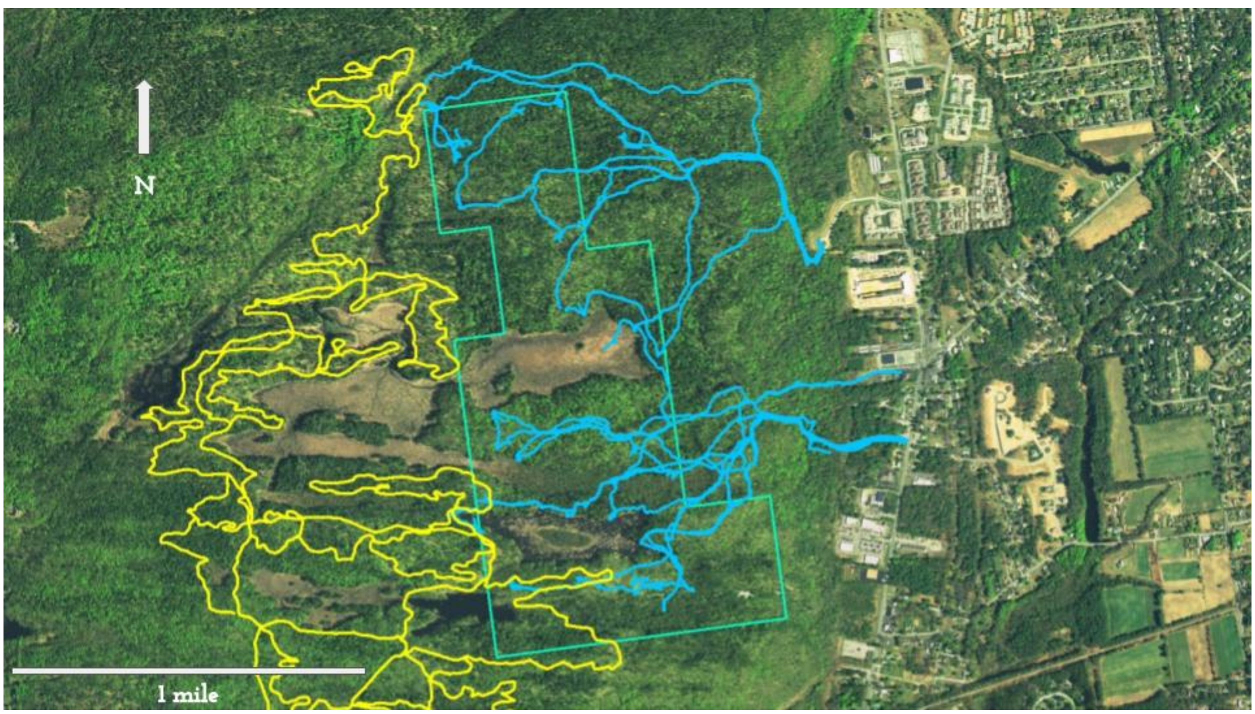

Figure A5. *Parcel and walking routes (blue) with SMBA's biking trails (yellow). This image shows how SMBA uses the land adjacent to the parcel, and some of their trails were even built on Skidmore's property. Image generated with ArcMap using NYS orthoimagery*.

Appendix B - Aerial imagery showing changes in development overtime.

All images are from Google Earth and have been modified in Powerpoint.

Figure B1. *Aerial image of the land including and around the Skidmore parcel.*

Figure B2. *Aerial image depicting creation of Hilltop Drive in 2015, shown within box.*

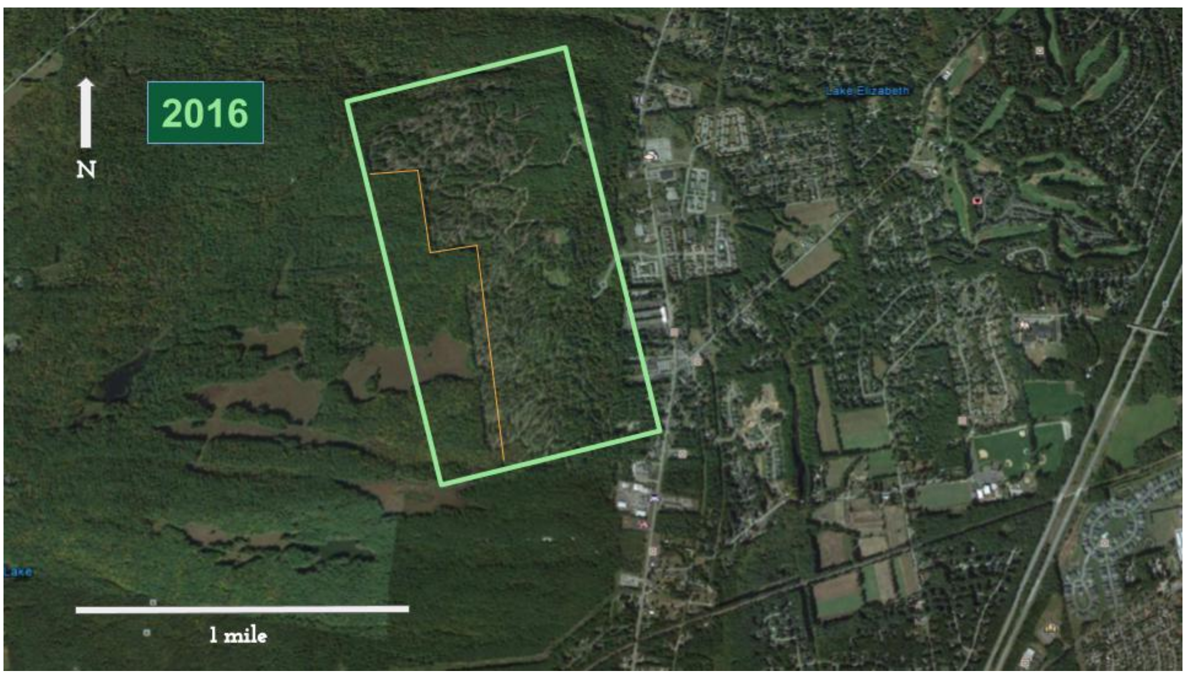

Figure B3. *Aerial image depicting logging event in 2016, shown within box.*

Appendix C - Data Tables and Figures

Plot	Land Cover CCAP	Trees #/plot	Biomass kg	Aboveground	Belowground	Leaf Litter	Forest Floor	Soil 0-10	Soil 10-20	Total
				----(C Mg ha-1) ----						
1	Mixed	134	16,309	108	26.6	2.8	2.8	66.5	40.1	247
2	Deciduous	125	12,671	84	21.3	2.0	2.4	107.7	41.3	259
3	Deciduous	153	14,782	98	24.4	4.2	2.2	56.6	40.3	226
4	Deciduous	186	12,393	82	20.8	5.0	7.0	193.6	70.8	380
5	Deciduous	136	17,439	116	28.2	2.8	3.6	63.0	36.5	250
6	Deciduous	132	14,034	93	23.3	2.8	4.6	87.3	37.6	249
7	Mixed	177	16,204	108	26.4	3.6	4.2	93.7	35.7	271
8	Deciduous	196	17,687	118	28.6	4.2	1.9	96.5	66.2	315
9	Deciduous	83	19,295	128	30.8	3.4	2.7	98.6	96.6	360
10	Deciduous	139	23,648	157	36.9	2.9	8.4	50.0	46.8	302
11	Deciduous	161	15,312	102	25.1	5.3	1.9	110.0	49.1	293
12	Evergreen	33	3,490	23	6.8	0.2	0.2	61.6	96.4	188
13	Evergreen	128	15,971	106	26.1	1.5	0.3	215.9	82.3	432
14	Deciduous	200	14,512	96	24.0	4.0	1.9	109.4	36.2	272
15	Deciduous	230	16,387	109	26.7	3.9	5.8	94.3	33.3	273
16	Mixed	96	16,822	112	27.3	2.5	2.3	72.9	47.2	264
17	Evergreen	143	17,008	113	27.6	2.4	3.5	174.7	78.8	400
18	Deciduous	53	18,334	122	29.5	3.8	3.0	59.0	21.8	239
19	Mixed	130	12,692	84	21.3	3.5	5.6	42.9	28.4	186
20	Mixed	104	14,016	93	23.2	2.1	2.5	39.5	26.4	187
21	Mixed	90	18,523	123	29.7	2.6	1.8	353.9	397.4	909
22	Mixed	208	16,790	112	27.3	4.8	3.0	117.0	118.9	383
23	Mixed	233	12,807	85	21.5	1.9	2.9	265.2	38.8	415
24	Deciduous	102	8,834	59	15.5	3.8	1.9	82.3	39.7	202
25	Evergreen	56	20,772	138	32.9	1.4	1.9	104.5	55.5	334
26	Mixed	198	11,264	75	19.2	2.3	2.9	91.7	49.0	240
27	Deciduous	148	13,249	88	22.1	2.4	3.9	155.4	44.2	316
28	Evergreen	141	11,244	75	19.1	3.1	1.6	110.6	55.3	264
29	Mixed	104	15,980	106	26.1	1.9	2.3	91.9	63.4	292
30	Deciduous	128	21,529	143	34.0	3.4	5.6	86.8	38.2	311
31	Emergent Herbaceous Wetland								854.4	854
32	Emergent Herbaceous Wetland								988.6	989
33	Forested Wetland								1,079.8	1,080
34	Open Water								72.2	72

Table C1. *Summary data for each of our plots' biomass, trees, and carbon storage. Wetlands only include the peat storage measurements with plots 31 and 32 both being in the northern wetland, 33 in the central wetland, and 34 in the southern wetland.*

Our Year	Forest Age	Forest Type	c_above	c_below	C_Soil	c_dead
	0	Deciduous	20.05	0	61.35	39.35
	5	Deciduous	22.25	0	61.35	27.25
	15	Deciduous	49.5	0	61.35	17.4
	25	Deciduous	75.45	0	61.35	13.95
	35	Deciduous	99.05	0	61.35	13.15
2017	45	Deciduous	119.6	0	61.35	13.65
2027	55	Deciduous	137	0	61.35	14.5
	65	Deciduous	153.2	0	61.35	15.6
2047	75	Deciduous	168	0	61.35	16.6
	85	Deciduous	181.7	0	61.35	17.6
2067	95	Deciduous	194.15	0	61.35	18.5
	105	Deciduous	205.45	0	61.35	19.3
2087	115	Deciduous	215.9	0	61.35	20.15
	125	Deciduous	225.35	0	61.35	20.85
	0	Mixed	33.9	0	66.9	30
	5	Mixed	30.6	0	66.9	23.6
	15	Mixed	45.6	0	66.9	17.2
	25	Mixed	68.6	0	66.9	13.6
	35	Mixed	90.8	0	66.9	12.1
2017	45	Mixed	110.8	0	66.9	11.6
2027	55	Mixed	128.1	0	66.9	11.6
	65	Mixed	144.6	0	66.9	12.1
2047	75	Mixed	157.5	0	66.9	12.6
	85	Mixed	169	0	66.9	13.1
2067	95	Mixed	179.3	0	66.9	13.6
	105	Mixed	188.4	0	66.9	14.1
2087	115	Mixed	196.3	0	66.9	14.6
	125	Mixed	203.2	0	66.9	15
	0	Evergreen	2.1	0	78.1	20.4
	5	Evergreen	9.5	0	78.1	16.5
	15	Evergreen	30.4	0	78.1	13.3
	25	Evergreen	46.5	0	78.1	11.4
	35	Evergreen	59.4	0	78.1	10.4
2017	45	Evergreen	71.1	0	78.1	10.1
2027	55	Evergreen	80.3	0	78.1	10.3
	65	Evergreen	88.4	0	78.1	10.5
2047	75	Evergreen	95.9	0	78.1	10.8
	85	Evergreen	102.8	0	78.1	11.2
2067	95	Evergreen	109.2	0	78.1	11.4
	105	Evergreen	115.1	0	78.1	11.8
2087	115	Evergreen	120.5	0	78.1	12.2
	125	Evergreen	125.4	0	78.1	12.5

Table C2. *Input values for the InVEST model to estimate the parcel's carbon sequestrations. Values are from Smith et al. (2006), where Deciduous values are an average of maple-beech-birch and oak-hickory (Table A2, A3), Mixed are oak-pine values (Table A4), and Evergreen are the white, red, jack pine tables (Table A6). All are clear cut tables.*

Appendix D - Qualitative Survey

Student Perceptions of Ecological Value

Q1 What is your current year at Skidmore College?

1st Year

Sophomore

Junior

Senior

Q2 What is your major and minor at Skidmore College?

Q3 What is your gender?

Male

Female

Other

Prefer not to answer

Q4 Ecosystems support human wellness and prosperity. From the provision of our food, to the regulation of our climate, humans are dependent on ecological systems. The benefits ecosystems provide to humans are known as ecosystem services. There are four main categories of ecosystem services: provisioning, cultural, regulating, and supporting. Based on the examples listed below, rank the ecosystem service you think is the most important (#1) to the least important (#4).

______ Provisioning services: providing food, water, timber and fiber

______ Cultural services: recreational, aesthetic, educational, and spiritual benefits

______ Regulating services: regulating climate, floods, disease, waste, and water quality

______ Supporting services: soil formation, photosynthesis, nutrient cycling, and species habitat

Q5 Ecosystem services are:

Those goods and services provided by ecosystems to humans

Dependent on ecosystem health and function

Services that provide both local and global benefits

All of the above

Q6 Out of all ecosystem services, only provisioning services are market benefits. Meaning the benefits provisioning services provide to humans are demonstrated in prices through markets. Cultural services, regulating services, and supporting services, are mostly "non-market" benefits. There is little representation of the economic value of "non-market" benefits because they have no prices. However, the benefits that "non-market" services provide are valuable. Hypothetically, how much of your annual tuition should be allocated by Skidmore College for non-market environmental services in the undeveloped land (owned by Skidmore), adjacent to the Daniels Road State Forest? Please enter a dollar amount:

Q7 Many of the benefits ecosystems provide to humans lack economic values ("prices") and are not traded in markets. It is debated if this lack of prices causes undervaluation of ecosystem services. Adversaries argue prices are harmful and turn the environment into a commodity. To what extent do you agree with the following statement: "I think dollar estimates of ecosystem services strengthen an environmental conservation agenda."
Strongly Disagree
Disagree
I do not know
Agree
Strongly Agree

Q8 Do you think the services the environment provides to humans should be represented by economic values?
Definitely, economic values can demonstrate the importance of ecosystem services relative to actions that cause environmental degradation
Maybe, but not all ecosystem services should be subject to economic values
Probably not, descriptions of ecological processes are more representative of the services the environment provides than economic values
Definitely not, using dollar amounts to frame the value of ecosystem services will undermine the intrinsic value of the environment

Q9 In general, there are three ways ecological economists price ecosystem services: market based, revealed preference, and stated preference. Based on the definitions below, which method do you think is the best representation of ecological value?
Market based: values from established markets
Revealed preference: values from people's spending behavior
Stated preference: values from individuals' stated "willingness to pay" for an ecosystem service
None of the above methods are suitable measures of ecosystem value

Q10 React to the following statement: "I think national measures of wealth should include ecosystem services as an economic sector in the gross domestic product."
Strongly disagree
Disagree
I do not know
Agree
Strongly Agree

Q11 Land management plans do not impact ecosystem services.
True
False

Q12 Forests provide which ecosystem service(s)?
Habitat for a variety of species
Timber, food, and fuel
Water purification and cycling
Carbon sequestration and climate regulation
All of the above

Q13 Where is Carbon stored in forests?
It's not stored in forests
In the trees and plants
In trees, plant, leaf litter, and soils
I don't know

Q14 Biodiversity loss can put critical forest functioning at risk, which can impair ecosystem services. True
False

Q15 Wetlands can help filter pollutants and improve water quality.
True
False

Q16 What are some societal benefits of wetlands?
During storm events, wetlands prevent or reduce the impact of severe flooding
Reduce costs for local residents/government's post-storm events
Wetlands act like a sponge, absorbing water during precipitation events and slowly releasing it after the event
All of the above

Q17 Wetlands are essential for the protection and improvement of:
Water quality
Fish and wildlife habitats
Capturing flood waters, and regulating water flow during dry periods
Climate change mitigation
All of the above

Q18 Plants that live in wetlands have the ability to:

Live longer than other terrestrial plants
Absorb excess nutrients from inputs, aiding in the filtration of water pollutants
Sense when severe storms are imminent
All of the above

Q19 Although wetlands represent a small area of land in the United States, what percent of all plant species live in wetlands?
17%
80%
31%
< 1%

Q20 Natural wetlands harm drinking water quality for nearby residents.
True
False

Q21 React to the following statement: "I understand the relationship between ecosystem health and ecosystem services."
Strongly Disagree
Disagree
I do not know
Agree
m Strongly Agree

Q22 If you would like to receive a follow-up report of the findings of this research, please write your email below:

SMBA Recreational Land Value
Q1 How many trips a year do you make to the SMBA trailhead located just north of the City of Saratoga Springs, at the corner of Clinton Ave and Daniels Road in Greenfield, NY?

Q2 How many miles do you drive to get to the SMBA trail head near Daniels Road?

Q3 If you are a mountain biker how much money do you spend yearly on bike maintenance?

Q4 List the activities you participate in when you visit the SMBA trail head near Daniels Road:(examples: mountain biking, hiking, bird watching, cross-country skiing)

Q5 Ecosystems support human wellness and prosperity. From the provision of our food, to the regulation of our climate, humans are dependent on ecological systems. The benefits ecosystems

provide to humans are known as ecosystem services. There are four main categories of ecosystem services: provisioning, cultural, regulating, and supporting. Based on the examples listed below, rank the ecosystem service you think is the most important (#1) to the least important (#4).
______ Provisioning services: providing food, water, timber and fiber
______ Cultural services: recreational, aesthetic, educational, and spiritual benefits
______ Regulating services: regulating climate, floods, disease, waste, and water quality
______ Supporting services: soil formation, photosynthesis, nutrient cycling, and species habitat

Q6 Many of the benefits ecosystems provide to humans lack economic values ("prices") and are not traded in markets. It is debated if this lack of prices causes undervaluation of ecosystem services. Adversaries argue prices are harmful and turn the environment into a commodity. To what extent do you agree with the following statement: "I think dollar estimates of ecosystem services strengthen an environmental conservation agenda."
Strongly Disagree
Disagree
I do not know
Agree
Strongly Agree

Q7 React to the following statement: "I think national measures of wealth should include ecosystem services as an economic sector in the gross domestic product."
Strongly disagree
Disagree
I do not know
Agree
Strongly Agree

Q8 In general, there are three ways ecological economists price ecosystem services: market based, revealed preference, and stated preference. Based on the definitions below, which method do you think is the best representation of ecological value?
Market based: values from established markets
Revealed preference: values from people’s spending behavior
Stated preference: values from individuals’ stated "willingness to pay" for an ecosystem service
None of the above methods are suitable measures of ecosystem value

Q9 Do you think the services the environment provides to humans should be represented by economic values?
Definitely, economic values can demonstrate the importance of ecosystem services relative to actions that cause environmental degradation

Maybe, but not all ecosystem services should be subject to economic values
Probably not, descriptions of ecological processes are more representative of the services the environment provides than economic values
Definitely not, using dollar amounts to frame the value of ecosystem services will undermine the intrinsic value of the environment

Q10 What is your age?
Under 18
18 - 24
25 - 34
35 - 44
45 - 54
55 - 64
65 +

Q11 What is the highest level of education you have completed?
High school
Some college, no degree
Associate degree
Bachelor's degree
Master's degree
Doctorate

Q12 What is your annual income?
$0 to $9,325
$9,325 to $37,950
$37,950 to $91,900
$91,900 to $191,650
$191,650 to $416,700
$416,700+
Prefer not to answer

Q13 What is your gender?
m Male
m Female
Other
Prefer not to answer

Q14 Thank-you for your participation in this research, all responses are confidential. If you would like to receive a follow-up report of the findings of this research, please write your email below:

Appendix E - Economic Analysis of Climate Change

During the initial stages of climate change more individuals will experience welfare loss, than welfare gain. Intuitively, an overall decline in global welfare would manifest as an aggregate cost. However, according to a 2.5°C temperature increase, the aggregate benefits of climate change experienced in the temperate zone outweigh the initial costs of climate change experienced in the tropics (Nordhaus, 2007; Tol, 2009). The temperate zone includes the United States, Canada, Europe, and Russia and represents the majority of the global economy of the world. While the tropics, including Central America, much of South America, Africa, and Polynesia, represents the majority of the global population.

The temperate zone will reap positive externalities from the initial stages of climate change (Tol, 2009). A moderate increase in carbon dioxide will reduce water stress in plants, allowing them to grow faster, thus stimulating the agricultural sector in the temperate zone (Tol, 2009). Additionally, a moderate increase in carbon dioxide emissions will result in warmer temperatures in the temperate zone and reduce heating costs (Tol, 2009). Meanwhile, in the tropics, temperature and rainfall variability will strain agricultural systems and cause economic instability. Low-income countries in the tropics will experience relatively greater impacts of climate change because of their lack of appropriate resources and reduced mitigation capacities (Tol, 2009). Relative to the economic powerhouses of the temperate zone, the costs of climate change in the tropics are marginalized against the USD (Tol, 2009).

The inequity between the welfare gain in the temperate zone and welfare loss in the tropics is misleading because climate tipping points will have a global impact. Tipping points in the Earth's natural systems cause abrupt, ecosystem change. When a tipping point is exceeded, a global environmental system collapses. Present ecosystem degradation heightens the role of uncertainty in the Earth's climate systems. Currently there is no overlap between valuation of ecosystem services and tipping points. The omission of tipping points in ecosystem valuation inhibits accurate projections of ecosystem value. Tipping points will result in ecosystem degradation, and ecosystem degradation will reduce the provision of ecosystem services. This likely decrease in the supply of ecosystem services is worsened by projections of population growth in the coming decades. As the number of people increases, the use of ecosystem services will also increase, making ecosystem services more scarce. When a good becomes scarce, it becomes more valuable. However, since ecosystems lack prices, the omission of tipping points excludes scarcity from economic value estimations.

The term, "tipping point" arouses broad speculation over the exact details of how, where, and when such tipping points will occur. Due to the non-linearity of climate systems, a lot of controversy in the literature surrounds how to predict critical threshold levels, quantify economic costs of tipping points, and identify when tips will happen (Alley et al., 2003; Schneider, 2004). Abrupt ecosystem change is a science accompanied with greater uncertainty than gradual ecosystem change. Unlike gradual ecosystem change, which allows ecosystems and economies time to adapt to changes, a tipping point will cause abrupt and immediate consequences.

In scientific literature, the term, “climate tipping points,” (Alley et al. 2003; Lemoine and Traeger, 2012; Rial et al., 2004) refer to the critical threshold at which a slight “tip” can alter the climate state: such a change may be abrupt and irreversible in nature. When the tipping point in an environmental system is reached, the future state of the ecosystem is qualitatively altered (Rial et al. 2004). Paleoclimatic data of large, rapid, widespread climate changes indicate the existence of past tipping points (Rial et al., 2004). Future tipping points may result in regional changes, such as lake eutrophication, or global changes, such as melting of the Arctic ice-sheet, and variation of weather patterns in El Niño-Southern Oscillation (Alley et al., 2013).