Agriculture Expansion in Franklin County Massachusetts: Analysis of Trade-Off Scenarios Between Agriculture Production and Ecosystem Services

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Agriculture Expansion in Franklin County Massachusetts: Analysis of Trade-off Scenarios between Agriculture Production and Ecosystem Services

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Abstract

Proposed food system developments outlined under the New England Food Vision (NEFV) require clearing over 6 million acres of forested land for conversion into agriculture. Franklin County, Massachusetts was used as a proxy for analyzing the greater New England landscape and to model the tradeoffs between ecosystem services of nutrient retention and carbon storage under five unique land-use scenarios. I hypothesize that because forested land cover provides the greatest capacity to sequester carbon and to reduce nutrient export, the value of these ecosystem services will not be outweighed by the projected increases in agriculture production.

I developed land-use scenarios based on USDA soil capability class and current land-use codes, current conservation restrictions as well as BioMap2 future conservation goals. I used the InVEST Natural Capital 3.1.0 geospatial modeling tools, including the Nutrient Retention and Carbon Storage modules, to quantify and value ecosystem services. Valuation of nutrient pollution was estimated under two different mitigation strategies including Wastewater Treatment (WWT) and Nutrient Management Plans (NMP) costs. The value of agriculture products was derived from the area converted into pasture or cropland from forests based on the NEFV dietary guidelines. I then estimated the wholesale value of vegetables, grass-fed beef, dairy, lamb and wool.

The model predicts that the combined value to society ranges from a net present value (NPV) of $99 million under the baseline land use pattern, to net loss of $307 million under scenario 5 with NMP costs. When considering the costs of the WWT the

net value nearly doubles, with a range of $184 million under the baseline scenario to $557 million under scenario 5. I conclude that maintaining the baseline land use pattern or scenario 4, which has limited agriculture expansion in the county, is the most sustainable option, and that all future conservation measures should be pursued to secure the value of ecosystem services for future generations. Pursuing the NEFV will cause wide scale destruction of ecosystem services that outweigh the value of increased local food production.

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Introduction

Increasing global human population has caused a subsequent increase in global food demand. According to the Food and Agriculture Organization of the United Nations (FAO), global food production must increase by at least 70% to meet the demands of the growing world population, which is projected to exceed 9.1 billion people by 2050 (FAO, 2009). In the United States, agriculture production will need to double by 2050 to meet the estimated demand. These production gains will need to be achieved while simultaneously addressing agriculture's overdependence on fossil fuels and its impacts on critical ecosystem services such as water regulation and filtration, and climate regulation through carbon storage. Nearly doubling U.S. agricultural production is a laudable and necessary policy goal, but significant questions remain as to where this increased production is going to occur and what tradeoffs it represents.

The development of a robust regional food system, as proposed in the New England Food Vision, has the potential to increase food security, increase available nutrition and stimulate local economies. Local food advocates have also proclaimed benefits for many assumed environmental benefits including the promotion of ecological habitat diversity, sequestration of carbon, nutrient retention, biodiversity preservation and water purification (Bommarco, Kleijn, & Potts, 2013; Edwards-Jones et al., 2008). Although many of the economic, social and public health benefits appear more intuitive, the complexity of environmental benefits requires greater examination. Many of these environmental benefits are assumed and not explicitly quantified or defined in relevant scientific terms by local food advocates.

What remains even more unclear is how a redeveloped and expanded regional food system will perform in the face of climate change and how it will impact ecosystem services on a landscape scale. Further, it remains unclear to what degree alternative farming practices and methods contribute to ecosystem services such as carbon storage and nutrient recycling. Decisions that result in land-use changes will have implied effects on the complex, hydrological, geochemical and biological systems that collectively function as tangible ecosystem services. Examining past changes in the landscape, and in particular the heavy footprint of agriculture, can inform our future aspirations of local food production, while grounding land-use decisions that preserve complementary and alternative environmental benefits.

Research Significance and Objectives

To better understand the consequences, a thorough analysis is needed to quantify the social and environmental benefits through tradeoff scenarios based on land use changes. This research project aims to discern where to best increase agricultural productivity within a varied landscape with mixed geophysical features and examine how these land use changes affect a suite of environmental services. The expected results will help inform policy decisions that achieve the dual objective of increasing regional agriculture output while protecting ecosystem services for future generations.

My primary research goal is to quantify the tradeoffs between ecosystem services resulting from projected landscape transformations caused by increased agriculture activity across a predominantly forested landscape. Further, I intend to examine the corresponding impacts on ecosystem services including carbon storage and nutrient retention through geospatial modeling of modified trade-off scenarios. I also seek to understand the implications of expanding regional food production by identifying and quantifying the environmental benefits associated with different land-use patterns. This analysis will highlight the most prudent ecological and sustainable land-use scenarios and geospatial features, while balancing increased food production. Examining how these changes affect ecosystem services will inform what the best regulatory framework should encompass and identify which lands should be conserved as forest or agriculture type.

By comparing alternative scenarios in which specific economic and ecosystem tradeoffs are compared, I will quantify various factors involved in managing the regional landscape jointly for optimal food production while balancing the tradeoffs of converting forested areas into agricultural production. The primary environmental benefits will include valuation of sequestered carbon and foregone nutrient pollution remediation costs. Economic considerations will include valuation of agriculture products including vegetables, dairy, grass-fed beef and lamb. Each scenario will undoubtedly need to reflect the management and conservation alternatives for the forested landscape as well as strategically relevant agricultural areas.

Background

In order to fully appreciate the potential of future food production in New England, one must consider the historical context between humans and their ability to manipulate the landscape. The region has undergone a series of transformations driven by anthropogenic forces beginning with Native American landscape manipulations through European migration and exploitation leading to wide scale deforestation. The region has also experienced population growth leading to urban and suburban sprawl and the ultimate decline of agriculture in the 20th century giving way to reforestation. Considering these factors is essential to understanding the region's food production capacity.

Historical Background of the New England Landscape

The indigenous cultures and subsequent European settlers that inhabited New England were intimately dependent on their ability to procure adequate nourishment. In this sense, local food production in New England is not a new concept, but a historical reality—a reality lost in the modern era of globalization and hyper capitalism. Prior to the modern era, food production was extremely local because of inadequate transportation methods and the challenges of preserving perishable products to overcome spatial and temporal constraints.

Since the last glaciation period, about 10,000 years ago, the New England landscape has undergone a series of transitions driven by natural and anthropogenic disturbances. Before European settlement forest composition was regulated by climatic changes, biological relationships and geophysical endowments as well as the effects of native peoples. Native Americans inhabited much of southern New England including the Algonquin speaking tribes of Mohican, Abenaki, Pocomtuc, Mohegan, and Wampanoag tribes. These native peoples frequently altered forest ecosystems to improve hunting conditions by clearing understory growth with controlled burns (Day, 1953; Russell, 1983). The intention of these slash and burn methods was to improve line of sight for hunting and to clear land for the cultivation of crops. The use of fire also increased species diversity through low level disturbance, which allowed for exploitation of successional species such as black raspberries, *Rubus occidentalis* (Cronon, 2011). Forensic evidence has suggested that several areas were passively managed by native peoples for several generations.

Land area used for pre-colonial cultivation was limited, but frequently occurred in tandem with seasonal migration to fishing grounds where fertile flood plains along the region's rivers could be exploited for primitive agriculture. Native peoples developed the "three sisters" technique to cultivate stalk corn, pole beans and squash. This was done by creating hilled mounds with the corn stalk providing structure for the climbing beans while the squash foliage provided ground cover for weed control. The leguminous beans provided nitrogen for the squash and corn. The cultivated areas were abandoned after one or two growing seasons as the semi-nomadic nature of the tribes did not allow for permanent cultivation or settlement (Cronon, 2011).

The stability of the forested landscape was dramatically altered after the arrival of European immigrants in the 16th century. By the middle of the 19th century Massachusetts forest cover was diminished to less than 40% of the total land area (Figure 1) and nearly the entire Native American population had been extirpated, assimilated or killed off. Most of the cleared land was converted into pastoral agriculture systems with only inaccessible marginal lands remaining forested. The peak of cleared land was witnessed by Henry David Thoreau in the 1850's, who described in his seminal essays on Walden Pond, the process of "succession" by which cleared land was abandoned, primary species are recruited, and the ecological process leading towards reforestation has been initiated. Since the time of Thoreau the main driver of reforestation in the region has largely been attributed to economic and ecological benefits from establishing agriculture infrastructure in the American Midwest and importing food and feed into New England (Donahue, 2007). As transportation costs were reduced with the development of the rail network, the region's exploding population became increasingly dependent on food imports (Ackerman-Leist, 2013). The creation and implementation of refrigerated cold storage only intensified this trend, as perishable foods could be transported across the continent and delivered to urban manufacturing centers throughout New England. The region's manufacturing industries provided economic opportunities for displaced farmers as well as subsequent waves of European immigrants, who required ever greater quantities of imported food. As a result of this great economic transition, farms and farmland across New England were abandoned (Figure 1).

Although the transportation of food, feed and fiber has occurred for several thousand years, only products that could not be produced locally were imported. Since the turn of the 19th century the economic phenomenon of importing food products has gradually been inverted to the point that nearly all of the Northeastern United States' food products are imported. Today, over 90% of the agriculture acreage footprint supplying the nutritional demands of New England's population are imported (Donahue et al., 2014). The heavy reliance on food imports has caused the abandonment of agriculture lands, which have been reforested. Reversing this trend towards more local food production will have significant impacts on the region's landscape and environmental services.

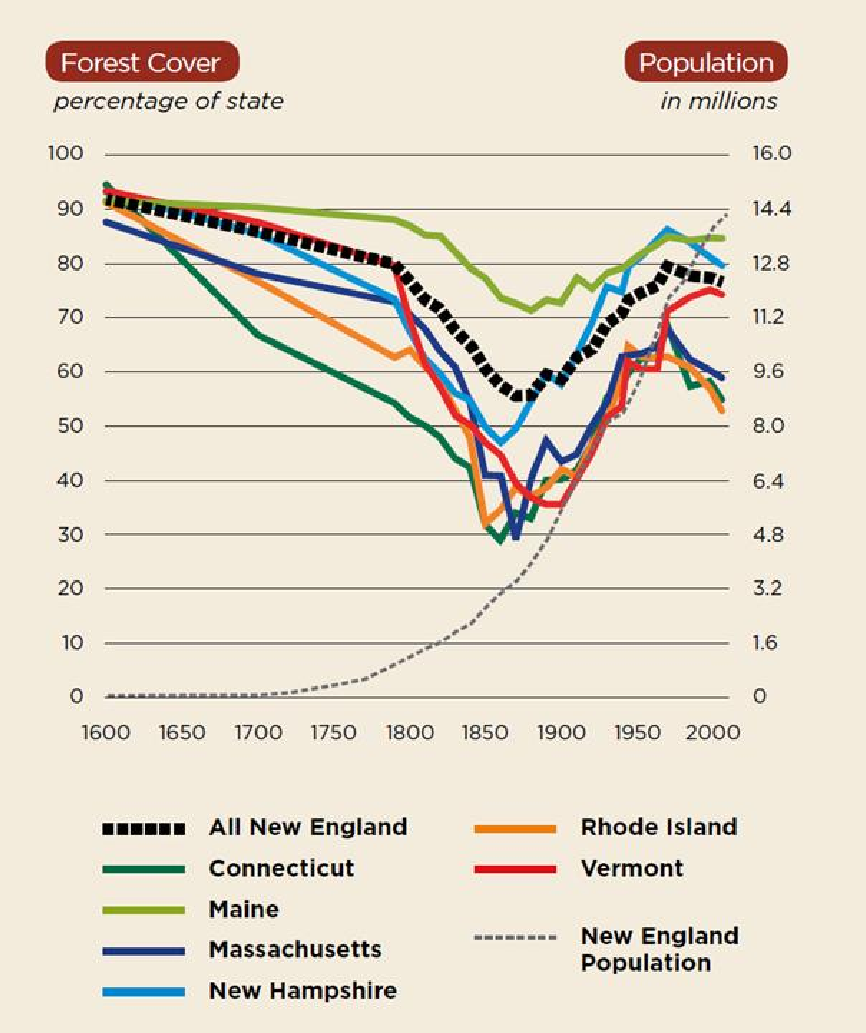


Figure 1. Changes in New England forest cover since 1600. Percent change in forest cover over the past 400 years for each New England State and total population change in millions (Donahue et al., 2014).

By the end of 19th century the vast impact from outsourcing of agriculture to other regions initiated a monumental land transformation in New England that reversed the movement toward peak deforestation in the 18th century, creating one of the most ubiquitous regional reforestation phenomenon ever witnessed in human history (Figure 1). The reforestation trend persisted until the latter half of the 20th century as abandonment of farmland continued, but was confronted by increasing pressure from urban and suburban sprawl. By the end of the 20th century suburban developments were deemed necessary to accommodate the region's burgeoning population beyond 10 million inhabitants. Although the trend of abandoning rural farmland has continued at a slower rate, the net impact on reforestation has been simultaneously counteracted by the increased rate of urban sprawl. The impacts of these changing forces has resulted in a net loss of forested land cover in Massachusetts since the 1970's (Foster et al., 2010). This period of urban sprawl is marked by a decrease in the total area of forested and agriculture lands. The impacts of the built environment are widely seen as a major threat to ecosystem services that the region's forest provide (Thompson & Broadbent, 2014; Thompson, Foster, Scheller, & Kittredge, 2011). The future of the New England landscape remains uncertain, because less than 30% landscape is protected from future development, which presents a serious threat to both forested and agriculture landscapes (Foster et al., 2010)

Increased Demand for Agriculture Output

Rapid global population growth throughout the 20th century has caused increasingly greater demands on existing agriculture systems (Ehrlich, 1970; Ehrlich & Ehrlich, 2009). Several projections by the UN forecast the global population to stabilize at 9.1 billion, but this may appear overly optimistic as there remains no reliable mechanism for slowing population growth in developing countries (Pimentel, 2004). The United States population is expected to grow from 319 million in 2014 to 417 million in 2060 (Colby, 2015). Massachusetts has not been immune to the global anthropogenic phenomena's of explosive population growth and urbanization; the state's population is expected to grow from 6.75 million in 2014 to 7.3 million in 2035 (Renski, 2015).

The USDA (United States Department of Agriculture) and FAO (Food and Agriculture Organization of the United Nations) have estimate that global food production will need to increase by at least 50% by the year 2050 to keep pace with growing demand. This is a monumental goal given the fact that nearly all of the prime farmlands are already in production and that certain regions are experiencing declines in production due to years of poor management and the effects of a changing climate. Global climate change is expected to compound the risk of world hunger. The Inter-Governmental Panel on Climate Change (IPCC) has projected increases in the frequency and intensity of crop failures and production shortfalls as regional climatic conditions become destabilized. Recent climatic abnormalities in major production areas (Russia in 2010, United States in 2012, and California in 2014) have caused global cereal grain shortages and food price increases (Porter et al., 2014). The IPCC also warns against the dangers from future climate volatility destabilizing production systems from increased occurrence of droughts, floods and pest infestations (Porter et al., 2014).

Further exacerbating the demand from population growth is the growing consumption of animal products in developing nations. Upward social mobility is often correlated with changing dietary habits that are more resource and energy intensive (Steinfield, Gerber, Wassenaar, Castel, & De Haan, 2006). Several studies estimate that agriculture production will need to increase by 60 to 110% in order to keep pace with population growth, but current growth rates in global agriculture production are inadequate to reach the FAO goal of 50% increased production by 2050 (Ray, Mueller, West, & Foley, 2013). Increasing global food production will require a concerted effort to expand production with improved technologies and methodologies as well as redeveloping agriculture infrastructure near urban population centers. This will inevitably require converting marginal lands into agriculture production to increase capacity and resiliency.

Industrial Agriculture in the 20th century

In response to the rapidly increasing global demand and the ensuing climatic changes the global industrial food system is miraculously expected to nearly double its production output. For the most part the industrial food system has been able to meet the nutritional demands of the global population explosion, but its upper limits are being questioned (Parker, 2011; Pimentel, 2004). Throughout the second half of the 20th century the United States and other developed nations developed an agriculture system that is excessively dependent on petroleum derived chemicals used for soil amendments, pesticides and energy inputs (Pollan, 2006). Not only are fossil fuels used to power farm equipment and transport agriculture products, they are the direct material from which synthetic fertilizers and pesticides are made. The global food system is also threatened by an increasing concentration of control of the food supply by multinational corporations (Verburg, Stehfest, Woltjer, & Eickhout, 2009). The concentration of food production into specific regions controlled by a limited number of corporate entities has likewise concentrated the risks caused by climate change because a limited number of crops and subsequent cultivars are widely employed. Monoculture agriculture systems compromise the genetic diversity of the crops and their inherent ability to adapt to changing conditions.

Currently the industrial food system is one of the largest contributors to climate change with the global food sector, contributing 18% of total greenhouse gas emissions (Steinfield et al., 2006); this amounts to more than all of the transportation sector combined (planes, trains, automobiles, ships, etc.). Fossil fuel resources initially enabled low cost long distance transportation of food products, but if petroleum prices increase food security risks will increasingly threaten regions that have become over dependent on imported foods.

Throughout the 20th century significant production gains were achieved through industrial agriculture's use of synthetic fertilizers and pesticides. Advances in agrochemical, botanical and bio-technological research that began in the 1940's have helped to stave off massive famines in what has been dubbed the "Green Revolution". These chemical inputs enabled significant increases in short term production outputs, but at the sacrifice of long term soil fertility; significant contributions to water and air pollution; and substantial emissions of GHG emissions (Diop, 2008). Increasing agriculture's dependence on chemical inputs to feed the growing population may prove to be a suicidal resolution, because of the detrimental effects on inherently complex biogeophysical systems that provide ecosystem services. Future advancements in agriculture science will ultimately be bound by this greater ecological context and consequences. Realizing the limits of current and future technologies and their relationship to ecosystem services will be essential for producing more food (Ray et al., 2013).

Public Health and Nutrition Related Diseases

Compounding the impacts of population growth and increasingly processed industrialized foods has been a correlated increase in Nutrition Related Diseases (NRD) (Myers & Bernstein, 2011). NRD now contribute to three of the top five causes of death in the United States: cardiovascular disease, cancer and diabetes. Obesity has been demonstrated to be a precursor for NRD and a causal factor for premature death. The rate of childhood obesity has reached epidemic proportions with significant consequences as obese children will experience challenges with weight management, psychological health, and a predisposition to NRD throughout their adult lives (Gillman & Ludwig, 2013; Popkin, 2004; Smoyer-Tomic et al., 2008).

The Harvard Medical School and the Harvard T.H. Chan School of Public Health have published the Healthy Eating Plate (HEP), which draws on decades of epidemiological research and clinical trials (Harvard University). The HEP recommended diet prioritizes vegetable and fruit consumption while minimizing red meat consumption. These recommendations can have a big effect not only on health but on GHG emissions as well, because red meat consumption is a major global contributor the climate change (Steinfield et al., 2006).

Prevention of NRD and obesity has been the focus of several policy initiatives aimed at reducing the public health costs associated with treating NRD (Popkin, 2004). Nationally there have been subsidies targeted at stimulating improved nutrition as part of the Supplemental Nutrition Assistance Program (SNAP). In addition to the SNAP subsidy for low-income families, the City of Boston has pioneered the Bounty Bucks program beginning in the late 1980’s that subsidizes the purchasing power of low-income residents receiving federal SNAP benefits. The Bounty Bucks program encourages residents with SNAP benefits to redeem their federal food subsidy at local farmer's markets where they are given a voucher that doubles their purchasing power. The policy has multiple benefits including reducing the cost of fruits and vegetables for low-income consumers, while increasing the sales volume of and cash flow directed towards local farmers. However, the scope and application of these innovative policy mechanisms are limited.

If the rate of NRD and obesity continue to increase as expected in the coming decades, additional resources and additional policy interventions will be required to mitigate these public health care costs. Connecting policy objectives aimed at improving the public health of low income residents while directing stimulus towards local farmers is an innovative approach that could be expanded. Supporting local agriculture is pragmatic and expedient means for improving the nutrition of the region's residents, but there should be qualifying criteria to ensure that the agriculture type and management practices are sustainable.

New England Food Vision

In response to concerns over public health, food security, population growth, climate change, environmental degradation and community resiliency there has been a groundswell of regional civic engagement fascinated with expanding local food production as a means to alleviating these global challenges (Ackerman-Leist, 2013). Several scholars from across New England have recently proposed the New England Food Vision (NEFV), which suggests that the region can and should produce half of the nutritional requirements for its population by 2060 (Donahue et al., 2014). The NEFV is meant to be an extension of the Harvard Forest’s Wildlands and Woodlands (W&W) vision for New England (D. Foster et al., 2010). Under the W&W vision, 70% of the total land area for the six New England States would be conserved as forest, with 10% of the land area conserved as protected Wildlands and the remaining 90% of the forested landscape managed for sustainable timber harvesting (D. R. Foster et al., 2010). Under this ideal W&W scenario 70% of the New England landscape would be protected as forested across the entire region with expansive urban sprawl in southern states and expansive (greater than 90%) forested regions in the northern states. By promoting conservation of the forested landscape the W&W vision ensures that environmental benefits will remain intact for future generations.

The NEFV suggests that the region can reasonably produce 50% of its own caloric food demands by 2060 without infringing on the forest conservation goals or ecosystem services outlined in W&W. The audacity of the NEFV can be realized when considering that the region currently only produces less than 5% of its caloric demands. The intention of the NEFV is to simulate public policy that will enable communities to become more resilient and sustainable in the face of climate change and the implications of regional population growth. The potential influxes of climate refugees, as well as the increased life expectancy of the aging population from improved health services, will likely cause rate increases that exceed the projected population of 17 million by 2060 assumed by the NEFV. These factors could reasonably cause positive feedback loops leading to increased life expectancy from the HSPH dietary recommendations and migration into New England. The demands from an increased future population will not only be realized from increased pressure for additional housing and food, but also increased demand for ecosystem services. An ever expanding regional population will likely place greater stress on the forested landscape to provide these services.

Although the W&W vision accounts for the vast majority (greater than 80%) of land area, there remains an additional 15% which can be utilized to expand regional agriculture. The available land, not protected as forest under the W&W, amounts to nearly six million acres. Reverting this land area back into agriculture production, as it was at the turn of the 20th century, would require clearing a significant area of the forested landscape. Currently agriculture activity in New England occurs on roughly two million acres, expanding it three fold to six million acres will have significant ecological impacts. Much of the environmental impacts will depend on where agriculture lands will be expanded, which crops will be grown, and the management techniques employed.

Under the proposed NEFV ideal Omnivore's Delight scenario, regional agriculture would produce all of the beef, poultry and dairy consumed by the region's population. This assumes a drastic change in the Standard American Diet (SAD), where excessive calories are derived from meat and dairy products, to the ideal diet outlined in the Harvard School of Public Health's Healthy Plate guidelines. Under the Harvard Healthy Plate guidelines meat and dairy consumption are significantly reduced in favor of a plant based diet. Assuming that the region's population will voluntarily adhere to the recommended nutrition guidelines defies the overwhelming trend in the SAD and the diet related diseases that are reaching epidemic proportions. It is a noble assumption that would significantly reduce in health care cost, environmental degradation and quality of life for New England residents. It should also be noted that if the dietary advice was widely accepted, it would cause a positive feedback loop of increased life expectancy and thus increased rates of population growth.

In order to produce all of the regional demand for meat, poultry and dairy, the NEFV also assumes that most of the region's grain requirements would be best grown in other regions of the continent. Importing grains for human consumption and as a feed stock reflects the landscape of New England, with its limited soil quality, hilly terrain and wet climate that is unsuitable for grain production. Importing grains is a necessary assumption, but it defers the environmental consequences of such practices to other regions. The NEFV correctly notes that imported feed stock will serve as an important fertility regiment for agriculture practices, but this fails to recognize that the initial source of fertility most likely originated from fossil fuels. The NEFV authors assume that imported grains will be produced with sustainable practices, but this overlooks how the vast majority of feed stock is produced in the United States and the vast volume of imported grains into New England. Feeding conventional grain products as part of a sustainable fertility strategy does not accurately reflect the true lifecycle of the grain crop and it petroleum inputs.

Climate change may prove to be serious roadblock for imported grains as persistent drought becomes more prevalent in the Midwest and fossil aquifers become depleted. Importing food products has been likened to importing virtual water (Brown, 2004; Brown, 2011), which is a reproduction of trading environmental services for cash. Our ecological deficit in the form of poor soil quality can be bought through the market transactions from regions whose farmers are willing to mine finite water resources for economic gain. Further, even if these imported grains are certified organic, there are no auditable or critically relevant criteria under the National Organic Program to safeguard procurement from hydrological abuses.

The NEFV assumes that all vegetables can be produced sustainably using season extension practices and shifting to seasonally appropriate diets. Growing all of the region’s vegetables has traditionally been limited by the seasonality of the region, which may prove to be harder with consumer expectations revolving around constant availability and heavy reliance from global imports. The NEFV authors correctly draw upon a wealth of long term studies that suggest organic and sustainable methods are more productive in the long term (Diop, 2008), but they fail to recognize the scope of sustainable agriculture in the context of current supply and demand requirements for feeding half of the region’s dietary needs. This is apparent when realizing that nationally less than 5% of the food supply is grown organically. Achieving the goal of 50% of caloric demands of the region would be an enormous task with conventional methods, never mind with more intricate production methods. Reaching this goal with environmentally prudent methods will require a higher degree of technical skill, ecological knowledge and innovative technologies-- all of which need to be developed and disseminated to the regions farmers.

The NEFV assumes a great expansion of sustainable agriculture practices, but it further fails to incorporate a substantial investment in agroforestry for food and feed stock. Donahue et al. (2014) discount agroforestry methods as unproven models, but these practices are a logical reconciliation of forest conservation and expanded agriculture production. Agroforestry may prove to be a viable and necessary link to close the gap between limited access to prime soils in the region and a vast expanse of marginal lands unsuited for mixed vegetable production. Agroforestry crops like hazelnuts could yield the greatest outcome of carbon storage while producing food fuel and feed. Failing to recognize agroforestry's potential also serves to discount agroforestry's historical role in New England for maple syrup and chestnut production.

One aspect of the NEFV vision that could prove to be the linchpin propelling the vision into reality or hindering its realization is that of energy efficiency and GHG emissions costs. The NEFV assumes that peak oil will be realized in the coming decades and that energy prices will increase and carbon costing will be implemented. The use of fossil fuels in agriculture has had by far the greatest impact in increasing output under conventional growing methods. Fossil fuels have enabled the green revolution of the 1970's, which have in part been responsible for the unrestricted global population growth. Higher energy prices will undoubtedly result in greater local food production because the economic advantage from economies of scale and remote production locations is negated by higher energy prices. Regional resiliency will depend on the region's ability to securely supply its own food as transportation costs rise. But this assumption uniformly affects all agriculture activities, even under sustainable agriculture methods petroleum is widely used in everything from fuel for tractors and farm implements; transportation of farm inputs and farm products; plastic materials used for covering greenhouses, drip irrigation, planting trays and food packaging. This does not include the tremendous amount of energy used in refrigerated cold storage and food processing. Although higher energy prices could ultimately prove to be an accurate assumption, it could also prove to be the tide that lifts all boats and hider the expansion of local agriculture that for the moment is mutually dependent on fossil fuels.

Under the NEFV agriculture activity would have to significantly expand from two million to six million acres, but they do not specify where and how these land will identified and for which agricultural activity they will be engaged in. This naively assumes that private land owners will make the most ecologically prudent land use decisions. In reality land owners will most likely make the most financially prudent land use decisions, which in recent history has been to develop the land into housing and commercial uses. Even if the land is converted into agriculture production, there is no guarantee that the production method satisfies the regional dietary recommendations of the Harvard Healthy Plate or is ecologically appropriate. The most likely scenario is that land owners will make land use decisions with short term or immediate financial returns. This would suggest that market forces and consumer demand will dictate which agriculture products are produced and recent trends in consumer demand revolve around the satiability of fat, salt and sugar. To achieve the NEFV robust policy mechanisms need to be developed to reshape nutritionally prudent consumer demand, as well as financial incentives for farmers to focus on ecologically appropriate methods and products.

Increasing the region's agricultural capacity as outlined by the NEFV is an ambitious goal, but several questions arise when considering the environmental tradeoffs of such development and what policy gaps need to be ameliorated. Accurate economic and scientific assessments are needed to provide the analytical framework for guiding adequate policy decisions aimed at addressing the region's potential and NEFV goals.

Several federal, state and municipal policy mechanisms are currently in place to foster agricultural growth by improving access to land, low interest loans for equipment and infrastructure needs and technical training (Bowell et al., 2014). Although there is a network of policy initiatives in place to maintain current farming operations, there remain considerable obstacles inhibiting the economic viability of small to medium sized farms in the region. Few of the smaller farm businesses are able to turn a profit and many of the new farmers have trouble accessing land. Most new farmers are leasing land with no long term contract, which disincentives the substantial and costly soil and infrastructure improvements that lead towards sustainability. Currently the Massachusetts state legislature is working on several policies to promote greater access to land and capital for aspiring farmers, but if significant increases in local production are to occur even more adventitious policies are needed.

Inevitably, global food production will have to increases everywhere to feed the increased demand, but this will depend largely on the cultural acquiescence, public policy atmosphere and ecological constraints of each particular community within the New England region. Developing regional and local food production systems beyond a marginal capacity will inevitably cause choices by local authorities and landowners that result in environmental sacrifices. Evaluating these sacrifices through environmental tradeoff scenarios will provide needed insight to the consequences of such decisions.

Ecosystem Services and Tradeoff Scenarios

The interdisciplinary science of valuing ecosystem services began in the 1970's and has evolved into a useful tool for guiding conservation efforts and landscape scale policy decisions (Vihervaara, Rönkä, & Walls, 2010). Ecosystem services refers to the goods and services provided by intact ecosystems that provide humans with well-being either collectively as social goods or individually as products and services (Seppelt et al., 2012). These goods and services are known as natural capital assets that collectively amount to essential life-sustaining public services such as clean air and water, as well as numerous economically important commercial goods such as wild fish stocks, petroleum reserves and harvestable timber. Many of these goods have a represented value within traditional markets, but their market-valuation inherently undervalues the true cost of production and overlooks the service value to society. This market failure of valuing public goods has led to the phenomena known as "tragedy of the commons" as described by Garrett Hardin (Hardin, 1968), where publicly owned resources often become over exploited for individual gain. North American ecological history is replete with examples of excessive exploitation by individual actors as a result of market failures and undervaluation of natural capital; such was the tragedy of the North American passenger pigeon, red wood forests, Atlantic salmon, bison, wright whale, and soil fertility on the Great Plains.

Furthermore, traditional capital markets are not capable of valuing less tangible, but more essential life-sustaining ecosystem services that include, but are not limited to clean air, clean water, carbon storage, species diversity, wildlife habitat preservation, pollination, drought tolerance, and flood prevention. Often the public value of these services exist as an externality to traditional market valuation and are thus underrepresented and undervalued as social goods, which makes these ecosystems vulnerable to excessive exploitation.

The valuation of ecosystem services is often estimated as the cost to replicate or reproduce the service through alternative means, such as constructing waste water treatment facilities to treat excessive nutrient pollution. In most situations the economic feasibility of reproducing ecosystems services would not be possible or would be very difficult to replicate and cost prohibitive. Establishing accurate valuation for intact ecosystems and their ability to provide public goods is not as straight forward as typical financial appraisals. First, analysis of the ecosystem's ability to provide services needs to be accurately measured and modeled; then value to society for replacing these services if lost or compromised are estimated. By assigning and assimilating the social value of ecosystem services into economic and financial analysis policy makers will be better able to conserve these services for future generations by making arguments that are fiscally responsible.

When considering future projects with landscape scale impacts it has become increasingly popular amongst policy makers to analyze the trade-offs between economic development and the acquired or forgone value of ecosystem services through trade-off scenarios (Brauman, Daily, Duarte, & Mooney, 2007). Scenario development should include the interests of multiple stakeholders to balance competing interest and inform management decisions. Management decisions affect the proportion, scope and relative mix of ecosystem function and the services provided. Balancing the short-term economic gains like that of increased rate of timber harvest against the long-term slow rate of carbon sequestration can analyzed through trade-off scenarios. Analyzing the ecosystem service trade-offs with spatial and temporal dimensions can inform better policy decisions and land use planning (Rodríguez et al., 2006). The complexity of the ecosystem services approach and its importance to conserving natural resources has led researchers at Stanford University to develop the Natural Capital InVEST software for modeling ecosystem services in collaboration with the University of Minnesota, The

Nature Conservancy, and World Wildlife Fund. The InVEST software package was used extensively in my research to model the ecosystem services of carbon sequestration and nutrient retention.

Climate Change and Carbon Sequestration

Imbalances in the global carbon cycle from anthropogenic sources have increased the concentration of atmospheric carbon pools. The scientific community has warned policy makers of the destructive consequences projected from atmospheric carbon dioxide (CO^2) concentrations of above 350 ppm. As of January 2015, the atmospheric concentration of CO^2 surpassed 400ppm and is expected to rise without aggressive policy measures to reduce emissions from fossil fuel combustion (Ciais et al., 2013). The scientific consensus is becoming increasingly clear that current emission levels risk triggering tipping points and irreversible impacts with significant economic and ecological costs.

Leveraging intact terrestrial ecosystems to sequester carbon from the atmosphere is one of the most attainable means of mitigating elevated atmospheric CO^2 concentrations. Terrestrial ecosystems including forests, wetlands and grasslands collectively store more carbon than the atmosphere (Lal, 2004). Photosynthesis drives carbon storage in forests, which contain four major pools, including below ground biomass, soil, above ground biomass, and dead organic matter. Accumulation and respiration of carbon in these four pools occurs at different rates, but the net effect of intact forest ecosystems is their capacity to sequester carbon. Early successional and younger forests accumulate carbon faster than older mature forests. Mature forests act as important carbon storage pools, but they cycle carbon closer to a state of equilibrium. Forest disturbance through natural or manmade events releases significant amounts of carbon into the atmosphere as well as disrupts the future sequestration capacity.

Managing forest ecosystems for their carbon storage potential will be critical to regulating the impacts of climate change. The reforested New England landscape has a substantial capacity for mitigating the effects of global climate change through carbon storage and sequestration. The value of the region's forests are expected to increase dramatically as policy makers adopt Reducing Emissions from Forest Degradation and Deforestation (REDD) and REDD+ frameworks. Recent research indicates that the region's forest has not yet reached carbon storage equilibrium and that above ground biomass accumulation could increase by over 65% by 2060 (Thompson et al., 2011).

Historically the threats imposed by climate change in New England have been measured in terms of forest composition and the influx of invasive pests (Foster, 2006). Climatic changes are expected to alter the frequency, abundance and timing of precipitation events, which could result in disrupted ecosystem services and ecological function. The changing climate is projected to increase the annual average temperature of the region's climate by 2.2 to 3.3 °C by 2041–2070. This increase in temperature correlates with an expected increase in precipitation over the same period by 4.7 to 9.5% (Tang & Beckage, 2010). The warmer and wetter climate that is predicted for the ensuing decades could impact both forest growth and the environmental services they provide. Regional climate instability and the associated impacts on agriculture production systems is poorly understood and considerable research is needed to understand the consequences.

Nutrient Pollution and Eutrophication

Nutrient runoff from agriculture poses a major threat to aquatic ecosystems and water quality. Non-point source pollution can lead to algae blooms and eutrophication. The two major pollutants are nitrogen (N) and phosphorus (P) affecting water quality from agriculture activity. Harmful toxic pathogens can also become a threat to drinking water supplies and overall water quality health from poorly managed manure systems. Conventional vegetable production requires heavy loading of N and P to achieve crop productivity expectations, although most of these nutrients are not absorbed rapidly which leads to runoff and loading. Aquatic plants are nutrient limited, which creates the potential for rapid biomass when low loads N and P are present. Rapid growth of phytoplankton and subsequent die-off can lead to reduced dissolved oxygen (DO) in benthic communities from aerobic decomposition. This process of algal blooms leading to hypoxia is known as eutrophication. In severe situations eutrophication can deplete DO throughout the water column causing fish kills and dead zones. Agriculture and urban runoff are the major causes of eutrophication as seen in the Gulf of Mexico (McCullough, 2013) and the Chesapeake Bay (Harding Jr et al., 2015; Scavia, 2015).

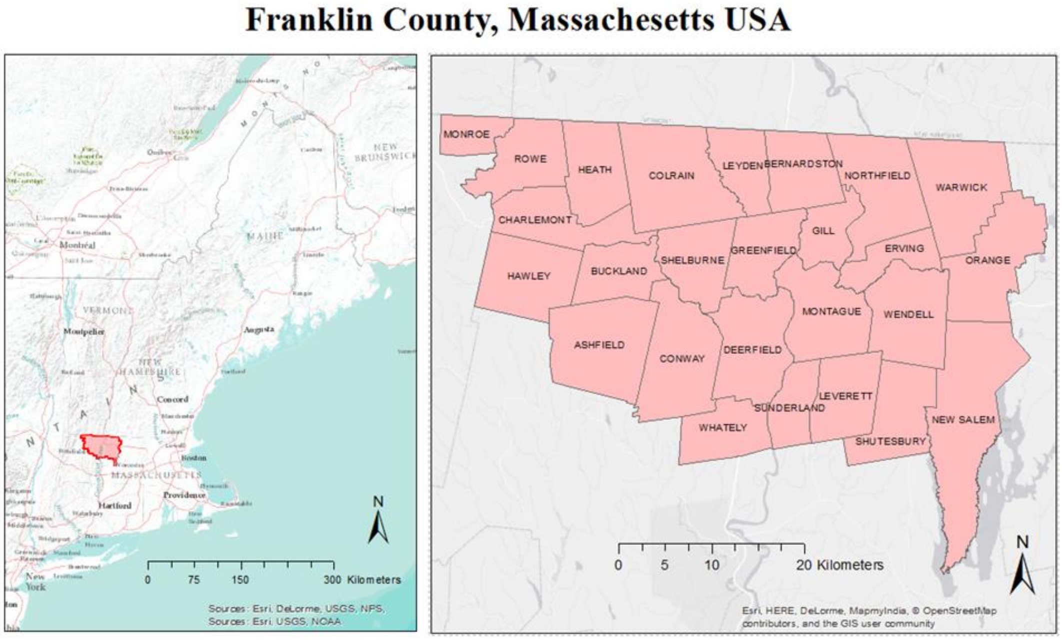
The USGS estimates that 14,638,096 kg of total N and 1,097,373 kg total P enter the Connecticut River Estuary annually. Fertilizer runoff accounts for 15% of the N load and 7% of the P load. Manure accounts for 9% of the N Load and 3% of the P load. The overall eutrophication level remains high from all tributaries in to the Long Island Sound including the Connecticut River Watershed (Moorman et al., 2014). Nutrient pollution from farming can also adversely affect vernal pools and wetland biodiversity (Sekar & Randhir, 2014).

The economic impact from eutrophication events can be immediate and long-lasting. Excessive nutrient loading can affect commercial fisheries, drinking water supplies, sanitation, recreation, biodiversity and species distribution and abundance (EPA, 2015; Mattson, Barletta, Godfrey, Wagner, & Aiello, 2003). Analyzing the impacts from land-use changes on ecosystem services requires both spatial and temporal dimensions. Assessing the economic and social value from regulating the effects of nutrient pollution by buffering and retaining excess nutrients is of great social and financial value to policy makers (Mattson et al., 2003).

Franklin County, Massachusetts

In response to the NEFV and the Massachusetts Food Plan released in 2015, the Franklin Regional Council of Governments (FRCOG) created the Franklin County Farm & Food System Project (FCFFSP) (Sloan, Praus, & Dechiara, 2015). The FCFFSP outlines the path forward to achieving the NEFV of farming in the county while integrating the interests of various stakeholders including conservation groups, farmers, food insecure families, academics, and policy makers. The FCFFSP acknowledges that in order to satisfy the NEFV over 16,000 hectares (39,537 ac) of additional farmland would need to be created. The report identifies the need for food sovereignty and food accessibility challenges for many of the counties rural residents. It also brings attention to loss of prime farmland to development and challenges new farmers face in accessing farmland. The FCFFSP provides an insight into the community’s desires and wishes for expanding local food production in the county. But the FCFFSP does not clearly outline where this expansion will occur or how it will impact other social goods like ecosystem services.

By analyzing the geospatial aspects of agriculture expansion in Franklin County I will be better able to understand the geophysical constraints and ecosystem impacts from local agriculture across the entire New England region. Exploring the ecosystem services impacts from expanded food production potential can best be analyzed by focusing on Franklin County as a subsample of the regional ecosystem, economics and agriculture infrastructure. By focusing on Franklin county, a detailed analysis will enlighten how, where and what types of agriculture are most appropriate given the physical features and natural endowments of the landscape. Figure 2 shows the relative proportion of Franklin County to the larger New England region.


Franklin County, Massachesetts USA

Sources: Esri, DeLorme, USGS, NPS,
Sources: Esri, USGS, NOAA

Esri, HERE, DeLorme, MapmyIndia, © OpenStreetMap contributors, and the GIS user community

Figure 2. Map of Franklin county. Reference to New England regional context and county level orientation with town names.

Geology of Franklin County

Like the greater New England region, Franklin County in Massachusetts is characterized by a diversity of geological substrates, topographic variations, soil conditions, preexisting agriculture infrastructure and it includes a similar distribution of land-use types. I chose to limit the study area to the county level because of the availability of data provided by the USDA and MassGIS. In addition, the USDA census data and agriculture metrics are routinely summarized at the county level. The USDA census data has been used to provide the context of agriculture activity for this project, while MassGIS datasets have been used to create scenarios of future land use.

Franklin County is located in the northwestern portion of Massachusetts (42°74'11.3"N – 42°31'35.5"S, 72°22'81.3"E – 73°02'34.0"W) and is the most rural county in the state. The county is comprised of 26 towns that combine to cover 724.5 square miles (187,642 ha). The climate of the county is cool temperate with an annual average of 131 precipitation days. The annual average rainfall and snowfalls totals amount to 46.2 and 59.2 inches respectively. The USDA plant hardiness zone ranges from 5a (-20 to -15 °F) in the Berkshire Hills to zone 6a (-10 to -5 °F) in the southern portion of the Connecticut River valley. Although there is subtle variation in climate across the county's topography, nearly 90% of the county is classified as hardiness zone 5b (-15 to -10 °F).

The Lower Connecticut River Major Drainage Basin encompasses the entire county and extends into the surrounding areas of southern Vermont, southern New Hampshire and Connecticut. The Connecticut River flows south from the Vermont – New Hampshire boundary and bisects the county into eastern and western portions, each with forested hills that run parallel to the Connecticut River in a north-south orientation. Franklin County's drainage network is divided into several sub-basins including the Deerfield, Millers, Connecticut, Westfield and Chicopee. These five sub-basins are further subdivided into sub-watersheds, of which I have identified 228 that are completely or partially incorporated by the county boundary. The five sub-basins were used as the primary unit of analysis in the nutrient retention modeling.

The county's geographical structure is dominated by features created through alternating periods of glaciation and more contemporary erosion and deposition patterns of the Connecticut River and its main tributaries. The rolling hills in the western portion of the county are mainly forested or in various stages or successive reforestation of abandoned pasture land, making Franklin County an ideal microcosm of appropriate scale to forecast implications to the broader New England region. The eastern portion of the County is encompasses the northern portion of the Pioneer Valley, which is dominated by extensive alluvial floodplains of extremely fertile soil.

The eastern portion of the county is dominated by the Worcester / Monadnock Plateau ecoregion characterized by forested hills, small farms and large conservation areas that include the Quabbin Reservoir, Montague Wildlife Management Area, Wendell State Forest, Warwick State Forest, Northfield State Forest and the Mount Grace State Forest among others. The Quabbin Reservoir covers over 39 square miles in the south eastern portion of the county and serves as a boundary between Franklin, Hampshire and Worcester counties. The reservoir was built in the 1930's by flooding three branches of the Swift River watershed and four towns in Hampshire and one town in Worcester County.

The western portion of the county is dominated by the southern extent of the Green Mountains and the Berkshire Highlands; these ecoregions are characterized by the foothills of the Berkshire subsection of the Appalachian Mountain Range. The western hills are mostly forested with fragmentation occurring from small towns and farms. The farm land in the western portion of the county is mostly managed as pasture due to poor soil quality on moderately steep slopes.

The central portion of the county is dominated by the Connecticut Valley, Berkshire Transition, and Vermont Piedmont ecoregions. Franklin County encompasses the northern portion of the Connecticut River Valley also known as the Pioneer Valley. Figure 3 shows the topography of the county and the broad flood plains located adjacent to the Connecticut River.

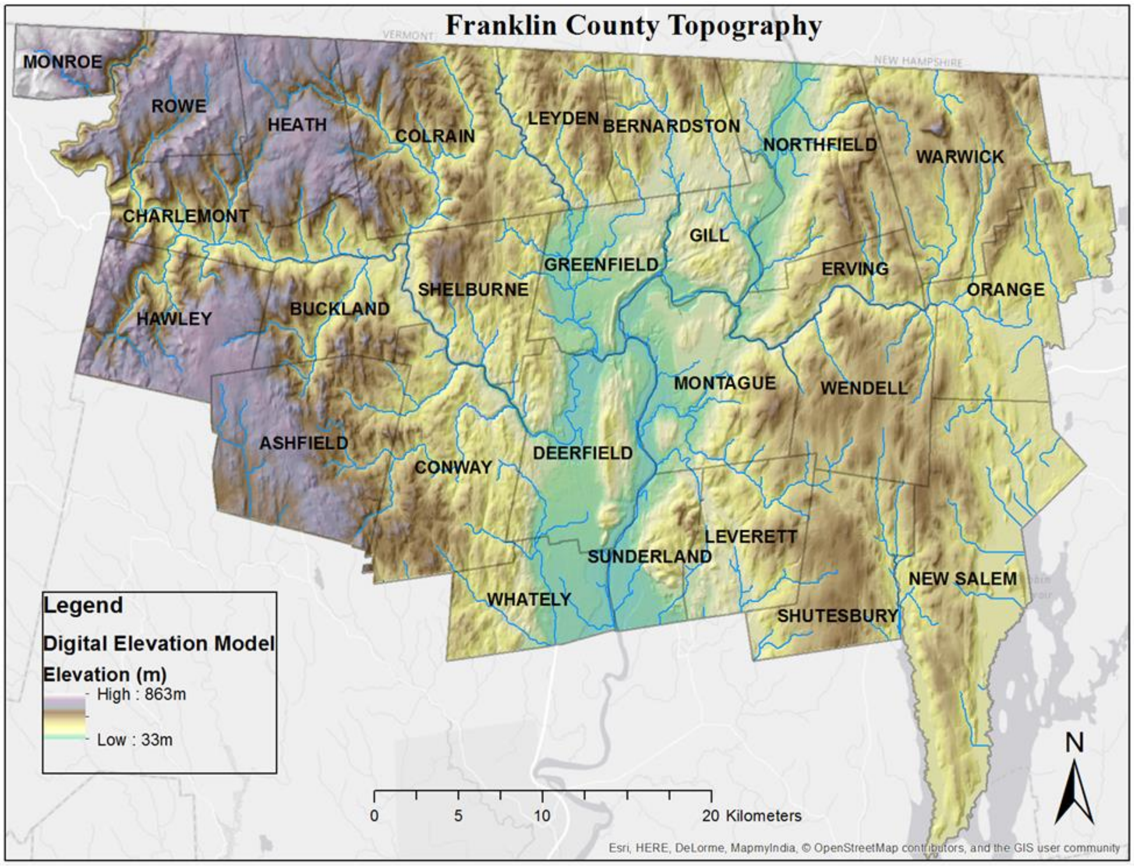


Figure 3. Map of Franklin county topography. Topography and elevation shown by color gradient ranging from 33 to 863 meters. River systems illustrated and town names are labeled.

The urban and suburban population centers, as well as row crop agriculture, are located predominantly in the fertile Connecticut River valley flood plains. According to the 2010 U.S Census, the county population was 71,221 and is located predominantly in several small cities and towns including Greenfield (17,565 pop), Montague (8,455 pop), Deerfield (5,096 pop), Turners Falls (4,620 pop) and Sunderland (3,696 pop). The population has remained relatively constant over the past decade.

Throughout the 18th and much of the 19th century the exceptional soil fertility crowned the Pioneer Valley as the most productive agriculture area in North America. During the 20th century much of the national agriculture production focused on the fertility of the Midwest and Central California. By the 21st century four main crops dominated the valley: tobacco, corn, potatoes, and various cucurbits (squash and pumpkins). In addition to crop production there is a large contingency of dairy farms and beef production.

Over the past two decades there has been a rejuvenated interest in the Valley's fertility and its regional significance for expanding food sovereignty. The efforts to revitalize the county's agricultural heritage and economic significance have been led by the Community Involved in Supporting Agriculture (CISA) organization. CISA's grass roots campaign has focused on fostering local demand for local agriculture products through its nationally recognized marketing efforts. The success of CISA's programs has played a pivotal role in revitalizing the agriculture community and consumption of locally produced food products in the county. Today, Franklin County continues to be one of the most productive agriculture counties in Massachusetts as well as one of the most developed agriculture economies in New England.

USDA Census Summary Data

Agriculture statistics for Franklin County were obtained from the USDA National Agriculture Statistics Service webpage using the Quick Stats 2.0 guided interface (USDA, 2016). Data was organized into summary tables for the census years of 2002, 2007, and 2012 with relevant information including total number of farms, crops volumes, crop acreage, livestock production and annual sales. This data was used to cross reference MassGIS land-use data as well as illuminate our understanding of current agriculture activity in the county.

According to the USDA National Agriculture Statistic Service (USDA, 2016), annual sales of livestock and animal products has remained relatively constant at $15 million dollars across the three census years of 1997, 2002 and 2012. It should be noted that animal sales in 2007 increased to $20.6 million dollars because of inflated dairy and meat prices driven by global financial insecurity, which impacted commodity futures of feed stock prices. Overall dairy production has continued to decrease across the decade as production capacity per cow has increased three fold from 7,000 lbs/yr/cow in 1960 to over 18,500 lbs/yr/cow in 2004. The general trend has shifted away from small family farms, which was the most common type of operation in Franklin County. During the 20th century dairy production shifted towards larger industrialized operations closer to corn and soy feed stock production west of New England (Blayney & Normile, 2004).

One notable trend observed in the USDA census data is the increase of local poultry production in the county. Poultry and egg sales have increased three fold from a combined total in 1997 of $500,000 dollars to $1,480,000 dollars in 2012. The increase in sales has been spread across a significant increase in the number of poultry operations, from 41 in 1997 to 134 in 2012. Although the increase in poultry production is significant it is has not been enough to outweigh the losses observed in the dairy industry. Figure 4 shows the decline of the dairy industry as improved efficiencies allowed for greater production per cow.

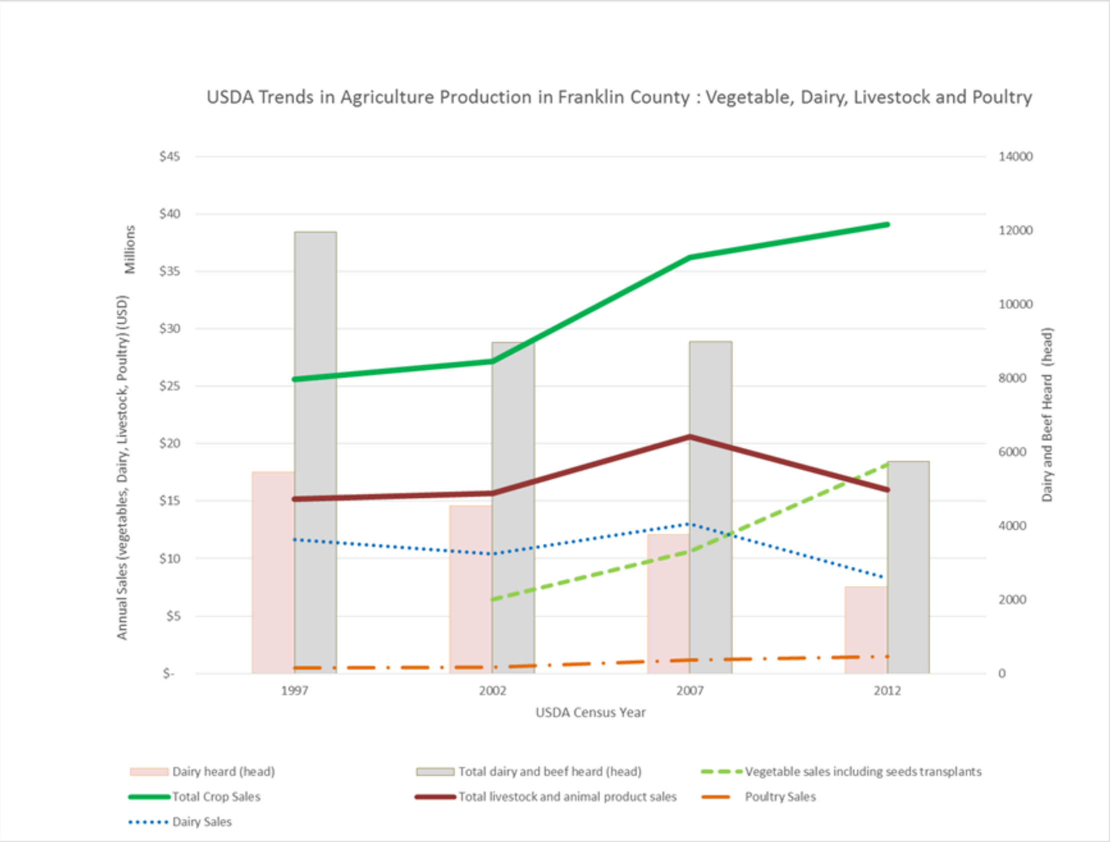


Figure 4. Graph of USDA trends of agriculture production in Franklin County. Note the annual sales volume for vegetables, total livestock, dairy and poultry. Dairy and beef heard size shown in columns. Census data for 1997, 2002, 2007, 2012 obtained from USDA quick stats (USDA, 2016).

The most recent census in 2012 illustrates the current amount of land actively farmed. Franklin County has 9105 hectares (22,499 ac) of cropland and 6,616 hectares (16,348 ac) of pasture. Of the cropland acreage, vegetable production accounted for 21.43% of the harvested acres (4,822 ac). Vegetable production includes both sweet corn and potatoes which compromise a significant portion of vegetable production acreage. The remaining 78.57% of the crop land acreage is largely devoted to low value feed crops including hay, haylage and grain corn. The trend of vegetable production has increased steadily in the county from 600 hectares (1,483 ac) in 1997 to 1,620 hectares (4002 ac) in 2012. Despite a decrease in overall active cropland from 7.5% (33,750 ac) in 1997 to 5%

(22,499 ac) in 2012 vegetable production has continued to increase. This trend of increased vegetable production is likely due to the phenomena of changing diets and the revitalization of local vegetable farms seen across the region over the same time period (Keough, 2012). Figure 5 illustrates the agriculture foot print by production type and Table 1 shows the combined value of crop and livestock sales for the USDA census years of 2002, 2007, and 2012.

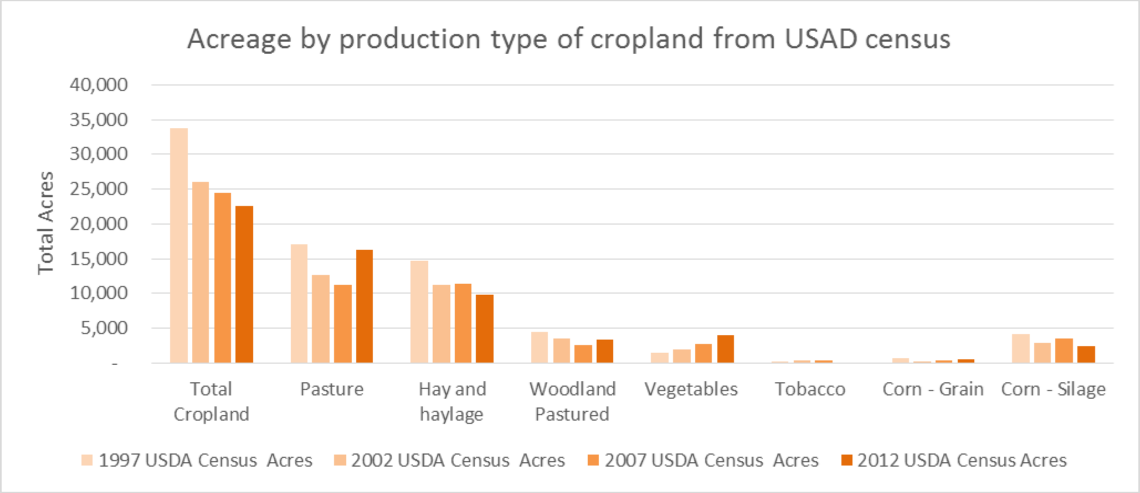


Figure 5. Trends in Franklin county agriculture acreage by production type from USDA census. Data captured from the USDA Quick Stats 2.0 for the years 1997, 2002, 2007, 2012. Note the increase in pasture and vegetable acreage in 2012 demonstrating a rejuvenation of the county's agriculture sector.

Table 1. USDA Census data of agriculture land allocation and total agriculture sales by major production type.

	1997 USDA Census		2002 USDA Census		2007 USDA Census		2012 USDA Census	
Agriculture Land Use	**Hectares**	**% of Total Area**	**Hectares**	**% of Total Area**	**Hectares**	**% of Total Area**	**Hectares**	**% of Total Area**
Total Farmland	32829	18.1%	30060	16.6%	32158	17.8%	36329	20.1%
Total Cropland	13658	7.5%	10521	5.8%	9886	5.5%	9105	5.0%
Pasture	6889	3.8%	5112	2.8%	4538	2.5%	6616	3.7%
Hay and haylage	5978	3.3%	4569	2.5%	4580	2.5%	3964	2.2%
Agriculture Woodlands	15601	8.6%	15693	8.7%	17432	9.6%	19138	10.6%
Woodland Pastured	1770	1.0%	1415	0.8%	1027	0.6%	1325	0.7%
Vegetables	600	0.3%	773	0.4%	1131	0.6%	1620	0.9%
Tobacco	101	0.1%	113	0.1%	126	0.1%	N/A	N/A
Corn - Grain	263	0.1%	108	0.1%	141	0.1%	234	0.1%
Corn - Silage	1643	0.9%	1194	0.7%	1420	0.8%	946	0.5%
Apple - Orchard		N/A	188	0.1%	205	0.1%	145	0.1%
	Number of active farms	**Sales $ (USD)**	**Number of active farms**	**Sales $ (USD)**	**Number of active farms**	**Sales $ (USD)**	**Number of active farms**	**Sales $ (USD)**
Total Livestock Sales (live animal and product sales)	319	$ 15,149,000	231	$ 15,705,000	320	$ 20,614,000	317	$15,992,000
Total Crop Sales	439	$ 25,588,000	390	$ 27,190,000	447	$ 36,230,000	448	$39,064,000
Total Revenue (sales)	679	$ 15,149,000	586	$ 42,895,000	741	$ 20,614,000	780	$15,992,000

Shifting dietary habits towards plant dominant diets is a major theme of the NEFV and was considered a major trend in the creation of scenarios for this project. Farms located in the Pioneer Valley of Franklin County are well suited to greatly expand their vegetable and fruit production because of economic advantages resulting from very high quality soils, adequate water resources for irrigation, and proximity to major markets of Hartford and New York City to the south, and Worcester and Boston to the east.

Although vegetable farms in the fertile Pioneer Valley are well suited for growth and expansion of markets, there is a finite amount of prime farm land for cultivated crops in the county. The reduction of total cropland in the county cannot be overstated as most of this land area has been irreversibly converted in to urban and suburban development. As noted in the NEFV, prime farmland will become a valuable social asset necessary to sustain the growing regional and global human population. The development of prime cropland is further confounded by the fact that 79% of the county land area has severe limitations for agriculture development. Most of the land area in the county has been allowed to be reforested since the 1830s because these marginal lands have lower soil quality or site limitations that make it best suited as pasture or to remain forested.

Long Term Conservation Structures

There are two primary categories for land conservation in Franklin County that include short term current-use conservation plans (Chapter 61, 61A and 61B programs) and long term conservation protections including Agriculture Preservation Restrictions (APR) and Conservation Restrictions (CR). Long term conservation structures are known as “easements of restriction” and are defined by legal deeds and documents that must meet the criteria outlined by the Massachusetts Executive Office of Energy and Environmental Affairs (EOEEA) under Article 97 Land Disposition Policy and Article 97 of the Massachusetts Constitution. Additional legal requirements are outlined under Sections 31-33 of Chapter 184 of the General Laws of Massachusetts. Private land is considered protected in perpetuity if it has development rights restricted by a deed, or if an APR or CR has been placed on it. Legal deeds restricting use can be developed and serviced by State agencies, land trusts, non-profits and local municipalities. State agencies with a legal restriction of land include the Department of Agriculture Resources (6,217 ha), DCR Division of State Parks and Recreation (3,696 ha) Department of Fish and Game (2651 ha) and the DCR Division of Water Supply Protection (252 ha).

Non-profit Land Trusts traditionally have two types of mechanism for conserving land: Conservation Restrictions (CR) and Agriculture Preservation Restrictions (APR). In Franklin County there are several non-profit organizations holding restricted use conservation easements. The two largest land trust include the Franklin Land Trust, which has conserved over 10,797 ha as of 2013, and the Mount Grace Land Conservation Trust, which has conserved a total of 11,590 ha, about half of which is located inside Franklin County boundaries.

There are over 2,195 parcels in the county classified under long term conservation agreements in Franklin County, of which 58,406 ha are conserved in perpetuity. The State owns a majority of these lands (37,589 ha) for conservation and water supply protection. Additional land areas have been conserved by private for profit entities (17,515 ha), local municipalities (4,429 ha) and land trusts (2,766 ha). These permanently conserved lands have been conserved through a variety of legal and financial frameworks designed for several intended purposes, some of which include: recreation, ecological conservation, water supply protection, agricultural landscape preservation, cultural-historical and flood control.

Short Term Conservation Structures

The Massachusetts Department of Conservation and Recreation (DCR) has designed three programs to incentivize conservation by rural landowners. The chapter 61 programs are defined by land use type and include: sustainable forestry (chapter 61), preserving active agriculture (chapter 61A), and preservation of open spaces (chapter 61B). All three programs discourage rural development while promoting conservation with financial incentives through reduced property tax rates. Land not enrolled in the chapter 61 program is enrolled under chapter 59 and assessed for its “highest and best use”. As of 2007 there were 640 parcels enrolled in the Forest Stewardship Program (Chapter 61) in Franklin County totaling 18,755 hectares.

To qualify for a chapter 61 forestry program the land area must be greater than 10 acres and the land owner needs to have a sustainable cutting plan created by a state certified service forester. The 10 year duration of the cutting plan duration is intended to guide the management of forest resources by private land owners. The program achieves the dual purpose of preserving the public good in the form of ecosystem services while providing annual income to landowners through sustainably harvesting forest products. The forested land can be managed for maple production, firewood production, and timber harvesting. The biggest benefit to landowners under the chapter 61 program comes from decreased property taxes because the enrolled land is assessed for its forestry use as opposed to the development value. When a land owner enrolls a parcel into the chapter 61 program the state places a lien placed on the property for 10 years. If the land owner deviates from the management plan they can be liable for back taxes at the higher rate (Van Fleet).

Similar to the chapter 61 forest conservation program, the chapter 61A program has been created to preserve active farmland from development. Under the 61A program private landowners can enroll parcels of land greater than five acres that have been actively managed for two years. Approved agricultural uses include land used for the production of fruits, vegetables, timber, animals, and animal feed, maple syrup and horticulture products. Permanent structures used for agricultural purposes can be included, but dwellings are excluded from the program and taxed at the standard rate. Forested areas may be included in the 61A program, but they must meet the same requires as the Chapter 61 program including a 10 management plan. Non-agricultural accessory land that is unproductive or unmanaged may be included in the 61A land area as long as it does not exceed 50% of the total parcel. Additional requirements for eligible land include that the landowner must demonstrate a minimum annual revenue of $500 dollars from the first five acres and $5 for every additional productive agriculture acre or $0.50 for every additional acre of managed forested land. Similar to the Chapter 61 program land owners are taxed at lower property rate. The assessed value of agriculture and forestry land enrolled in the Chapter 61 – 61A program is established annually by the Farmland Valuation Advisory Commission.

The correlating conservation for open space and recreation areas is known as chapter 61B. Under the 61B program landowners must have a minimum of five acres to enroll in the program. The land must fall into one of two categories, either "open space" or "recreation". Open space is defined as, "land maintained in a substantially natural, wild, or open condition; land maintained in a landscaped or pasture condition; or managed forest an approved 10-year forest management plan" (Van Fleet). The second land category under 61B pertains to land suited for recreation, but only if the recreation activity does not significantly impact the environmental integrity of the land. Approved recreation activities include, but are not limited to hiking, hunting, camping, horseback riding, fishing, skiing, swimming, picnicking, etc. Public access is required for recreation land or at least that it be accessible to members of a non-profit group, but public access is not required by the land owner under the open space category. The valuation of the land in chapter 61B is assessed at its recreation value, but not exceeding 25% of its assessed value under chapter 59.

Future Conservation: BioMap2

Future conservation goals have been spatially organized and focused in the Massachusetts BioMap2 datalayer published by the Massachusetts Natural Heritage & Endangered Species Program and the Nature Conservancy's Massachusetts Program. BioMap2 was created by a consortium of scientists using sophisticated modeling techniques that incorporate the dynamics of conservation biology and species-specific ecology. The intent of BioMap2 was to focus future conservation efforts; as described by the Massachusetts Department of Fish and Game:

> BioMap2 is designed to guide strategic biodiversity conservation in Massachusetts over the next decade by focusing land protection and stewardship on the areas that are most critical for ensuring the long-term persistence of rare and other native species and their habitats, exemplary natural communities, and a diversity of ecosystems. BioMap2 is also designed to include the habitats and species of conservation concern identified in the State Wildlife Action Plan. (Commonwealth of Massachusetts, 2010).

BioMap2 consists of the two main categories of Core Habitat and Critical Natural Landscapes that collectively identify conservation on multiple scales including species, ecosystems and landscapes. The Core Habitat includes intact forest ecosystems, priority natural communities, high quality aquatic and wetland habitats, and habitats for rare and vulnerable flora and fauna species. The Critical Natural Landscape includes large unfragmented landscape blocks, transitional lands that buffer wetlands and aquatic habitats (Commonwealth of Massachusetts, 2010). Figure 6 shows the spatial distribution of Core Habitat and Critical Natural Landscapes.

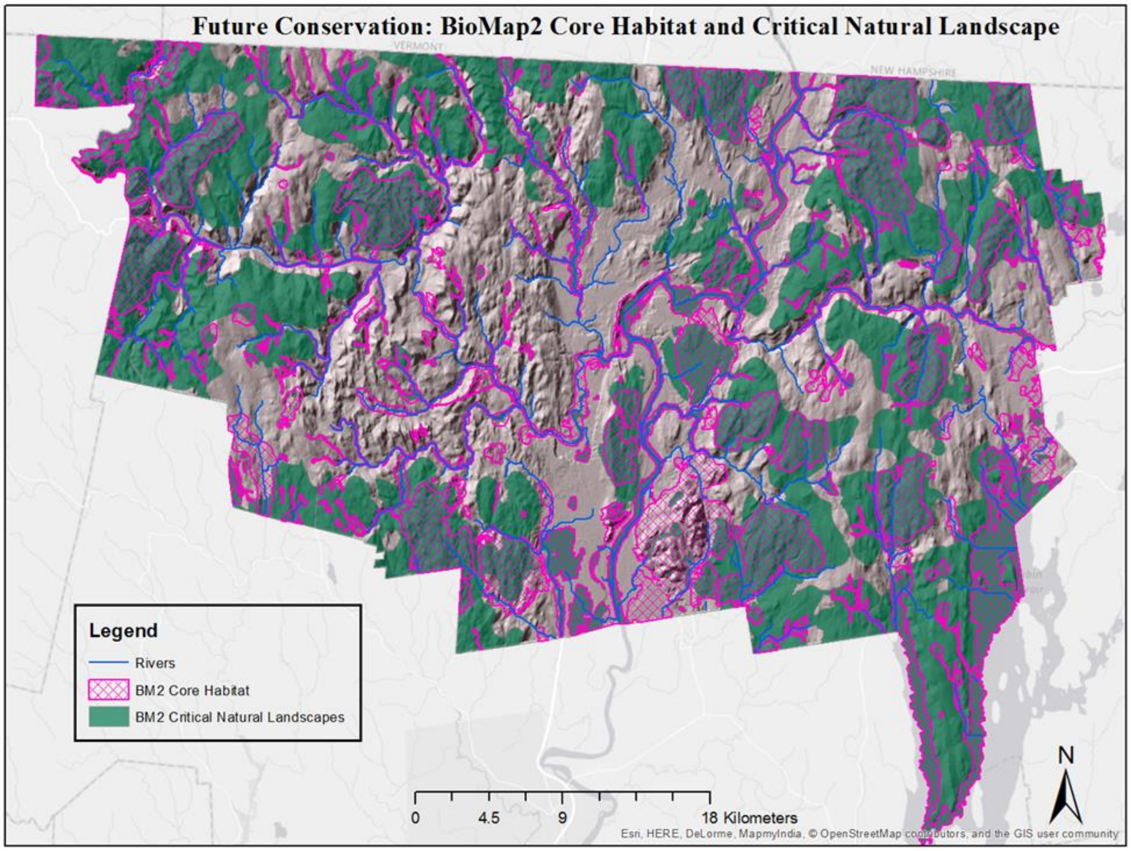


Figure 6. Map of BioMap2 major features. Geospatial display of BM2 Core Habitat, Critical Natural Landscape and Rivers.

NRCS Soil Capability Classes

The USDA National Resource Conservation Service (NRCS) Soil Survey capability class indexed ranking for soil quality was used as the primary criteria for developing scenarios of agriculture expansion. The soil capability class system provides a relative ranking, which demonstrates the degree of difficulty for the conversion of land areas into conventional cultivated cropland. The soil capability index system was created from the NRCS SSURGO database of soil properties collected over the past century, which assigns a feasibility score based on soil composition, soil depth, drainage, slope, existing infrastructure, pH, salinity, soil texture, precipitation, risk of flooding, and existing land cover (NRCS, 2014; USDA, 1967). For this project the index of Non-Irrigated Land Capability Class system was used.

Soil capability classes 1 and 2 are generally considered prime farm lands. Capability classes 1 and 2 have slight to moderate restrictions with adequate soil moisture, gentle slope, and climatic conditions. The soil depth, structure and composition in classes 1 and 2 are the most conduce for growing row crops. These prime soils allow for a wider variety of crops to be grown while requiring the least amount of site modifications.

Soil classes 3 and 4 have moderate to severe restrictions that, "reduce the choice of plants or require special conservation practices and management" (NRCS, 2014). These limitations include moderately steep slopes, erosion control challenges and water drainage problems. These areas may have moderately inadequate soil structure, composition, shallower soil depths and lower fertility that make it more difficult to manage for agriculture production. Although classes 3 and 4 could reasonably be used for cultivated crops they would likely require more inputs and site modifications as well as more intensive management. It is assumed that these land areas are less capable of growing cultivated crops and are best suited as pasturelands.

Classes 5 and 6 are considered to have severe to very severe limitations. These areas present significant challenges to cultivating crops and most likely require considerable site modifications including the construction of drainage, erosion and terracing systems. These areas also require significant soil structure amendments to improve fertility and prevent crop damage. The NRCS suggested the best use for these severely limited areas, "that make them generally unsuited to cultivation and that limit their use mainly to pasture, rangeland, forestland, or wildlife habitat" (NRCS, 2014). Soil classes 7 and 8 are considered to have major limitations making them not feasible for agriculture activity. These classes represent land areas with severe to extreme limitations for agriculture expansion and are characterized by steep slopes, very poor soil quality, severe erosion and pervasive hydrologic challenges.

Research Questions, Hypotheses and Specific Aims

In order to better understand future capacity of agriculture production we first need to fully understand current land-use patterns and agriculture capacity in Franklin County. An analysis of ecosystem services across Franklin County has not been published and the value of these services is unknown. We also do not know the agricultural production capacity of the county and its potential contributions to the NEFV goals. This analysis will ask first, what is the most ecologically prudent land use preference leading towards sustainability? This is in contrast to historical land-use decisions that ask only, what is financially optimal now? Once the most ecologically sustainable scenarios are identified, economic and financial analysis can be used to suggest public policy instruments that could be used to expand local agriculture while preserving robust ecosystem services, and maximizing public benefits.

Developing trade-off scenarios based on soil quality parameters will enable exploring realistic expansion regimes. Future expansion is limited to geophysical endowments, which need to be identified and mapped. By identifying suitable lands we will be able to construct scenarios that balance current and future conservation goals. Constructing scenarios that spatially recognize conservation areas will inform our current understanding of how conservation impacts ecosystem services and their value to society for providing these services.

Adding historical data to the geospatial mosaic will add a temporal dimension that will inform the current understanding of how the landscape has changed since the era of peak deforestation in the 19th century. Reconstructing historical land use patterns will provide some insight into how much agriculture production the landscape was previously capable as well as the consequences on ecosystem services. Assessing a historical scenario will also inform a better understanding of how the landscape could be managed into the future.

Perhaps the most interesting research question we will attempt to answer stems from determining the county's agriculture capacity and balancing expansion impacts on ecosystem service benefits. By identifying the thresholds where agriculture can be expanded without severely compromising ecosystem services benefits is a necessary step for achieving a sustainable food system. Calculating the trade-offs from expanding agriculture activity between various scenarios will provide a detailed analysis of which land management types would provide the greatest output of food while not infringing upon environmental resources. Identifying these thresholds of expanding local agriculture will significantly contribute to the understanding of what sustainable agriculture looks like on a landscape scale.

Hypotheses

My initial hypothesis emphasizes the importance of forested land areas for providing ecosystem services. I anticipate that land-use changes leading to pasture from forest will have significant negative impacts on both nutrient retention and carbon storage. Further, we expect the transformation from forest to cropland to have the greatest negative impacts on ecosystem services. The conversion of marginal lands is expected to have minimal benefits for increased agriculture production, while significantly detracting from nutrient retention.

I further hypothesize that the volume of land area classified as forest directly correlates with increased carbon storage, and expect modest increases of stored carbon in pastureland over cultivated cropland because of respiration caused by active tilling. Maximizing the forested land area is expected to provide the greatest environmental benefits, but the lowest economic return from agriculture output. Scenarios that emphasize conservation are expected to be the most advantageous when considering the social value of nitrogen pollution and sequestered carbon. The social value from ecosystem services is expected to far outweigh the financial benefits from increased agriculture activity.

Specific Research Aims

I first aim to identify strategically relevant land areas with appropriate soil types for agriculture production in Franklin County Massachusetts based on the NRCS soil classification. A focus on soil attributes identifies the spatial distribution of lands suitable for cropland or pasture conversion. The preliminary analysis will include spatially identifying land areas capable of crop production as well as sub-prime marginal lands best suited as pasture or forestland. Quantifying the agriculture production capacity of the county as defined by soil type, slope, elevation, vegetation cover and current land-use will require overlaying the geospatial data layers into a framework that reclassifies the landscape based on geophysical determinants. Once these variables are quantified, the geospatial framework will be the basis for analyzing land-use changes through a range of trade-off scenarios that depict relative variations between the ratio of forest, pasture and cultivated cropland.

Modeling changes to the geospatial mosaic will enable an assessment of land-use changes that specifically contribute to carbon storage and nutrient retention across the landscape. By quantifying the capacity of these ecosystem services, I will then be able to model how much crop and pasture land expansion impacts ecosystem services. More specifically, this modeling design will attempt to answer several questions including: what are the implications to ecosystem services in terms of N and P pollution (kg/watershed) and forgone carbon sequestration Mg/C) resulting from the conversion of forested lands into actively managed pasture or cropland? I will also explore the valuation of these services, including the cost to remediate excess nutrient loading and the market value of sequestered carbon. I will conduct a cost benefit analysis that balances forest conservation with increased agriculture production, by estimating the value of agriculture products against the social value of ecosystem services.

Chapter II

Research Methods and Design

Research methods were designed to utilize the InVEST natural capital project software for modeling ecosystem services for carbon sequestration and nutrient retention. Trade-off scenarios were created by projecting land-use changes that reflect various levels of agriculture buildout under the assumptions and context of the NEFV. The 1830s scenario was designed to illuminate the ecosystem service impacts from landscape scale deforestation due to a predominately pastoral landscape. Two different valuation methodologies were analyzed for remediation of nutrient pollution under each scenario. The final analysis balances the value of agriculture products against the social value of ecosystem services.

GIS Data

The coordinate system used was NAD83 (horizontal) and the projection was Massachusetts State Plane Meters and NAVD88 (vertical) Meters. Criteria for establishing the scenarios included historical and current geospatial data published by MassGIS and the Harvard Forest. Geospatial parameters of analysis included soil quality, slope, current and future conservation plans, as well as historical land-use data provided by the Harvard Forest (Hall, 2002). I used several digital data sets obtained from MassGIS including: a Digital Elevation Model (DEM 1:5,000) (2005), Crop Evapotranspiration and Potential Evaporation Grids (2005), Land Use (2005), Drainage

Sub-basins (2007), County Boundaries (2014), NRCS SSURGO-Certified Soils (2012), Protected and Recreational OpenSpace (2015), BioMap2 (2011), Massachusetts Forest Stewardship Program Properties (2002)(MassGIS, 2015). Other data sources included the Harvard Forest's 1830s land use map (2002) and the 1830s road network map (2002)(Hall, 2002).

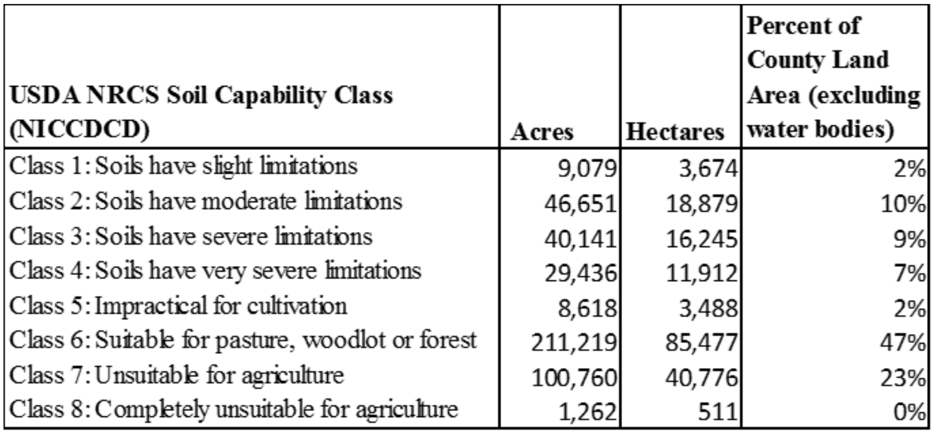
The MassGIS Land Use (2005) data layer was used to construct the baseline and first four scenarios. The data were prepared for MassGIS by private contractor Sanborn using semi-automated methods for classification and coding. The Land Use datalayer was created using 4-band digital ortho imagery captured in April 2005. The minimum mapping unit is generally 1 acre for non-urban areas. The data was edited and verified for accuracy through an extensive verification process by Sanborn with ancillary input data and onsite field verification provided by MassGIS. At the time of this project the MassGIS Land Use data set was verified and validated by MassGIS and was perceived to be the most accurate data publicly available for Franklin County.

NRCS Soil Capability Class Distribution

NRCS Soil Survey Data was used as the primary variable for selecting land areas with appropriate attributes leading to the possibility of agriculture development. The NRCS SSURGO-Certified Soils datalayer for Franklin County has 32,402 polygons (vector data points) and each polygon has numerous attributes, of which the Non-Irrigated Capability Class Index was utilized as the primary variable to spatially identify land areas suitable for agriculture development. I assumed that the relative ranking indicated by the soil Capability Class index layer to be the best single variable upon which to select land areas for future agriculture expansion based on site attributes in this project.

Based on the NRCS soil index, almost 70% of the county is classified (class 6 and 7) as marginal sub-prime land. Nearly half of the county land area (47.2%) is classified as soil class 6 (85,477 ha), and soil class 7 amounts to 22.5% (40,776 ha). These marginal lands are predominantly located on steeper terrain typical of the eastern and western portions of the county with mostly forested land cover and limited agriculture potential except for use as pasture. Soil capability class data was spatially mapped, as well as organized into summary Table 2.

Table 2. Franklin county land allocation by NRCS soil capability class.



USDA NRCS Soil Capability Class (NICCDCD)	Acres	Hectares	Percent of County Land Area (excluding water bodies)
Class 1: Soils have slight limitations	9,079	3,674	2%
Class 2: Soils have moderate limitations	46,651	18,879	10%
Class 3: Soils have severe limitations	40,141	16,245	9%
Class 4: Soils have very severe limitations	29,436	11,912	7%
Class 5: Impractical for cultivation	8,618	3,488	2%
Class 6: Suitable for pasture, woodlot or forest	211,219	85,477	47%
Class 7: Unsuitable for agriculture	100,760	40,776	23%
Class 8: Completely unsuitable for agriculture	1,262	511	0%

The spatial distribution of soil classes 1, 2, and 3 illustrates the concentration of prime farmland in the Connecticut River Valley. Most of the prime soils are either classified as class 2 (18,879 ha) or class 3 (16,649 ha). Currently most of soil classes 1 and 2 are in active production according to the 2005 MassGIS Land Use data layer. Expansion of agriculture onto prime farmland is limited within the county and could only be considered by converting lands with higher relative rankings. Figure 7 illustrates the spatial distribution across Franklin County.

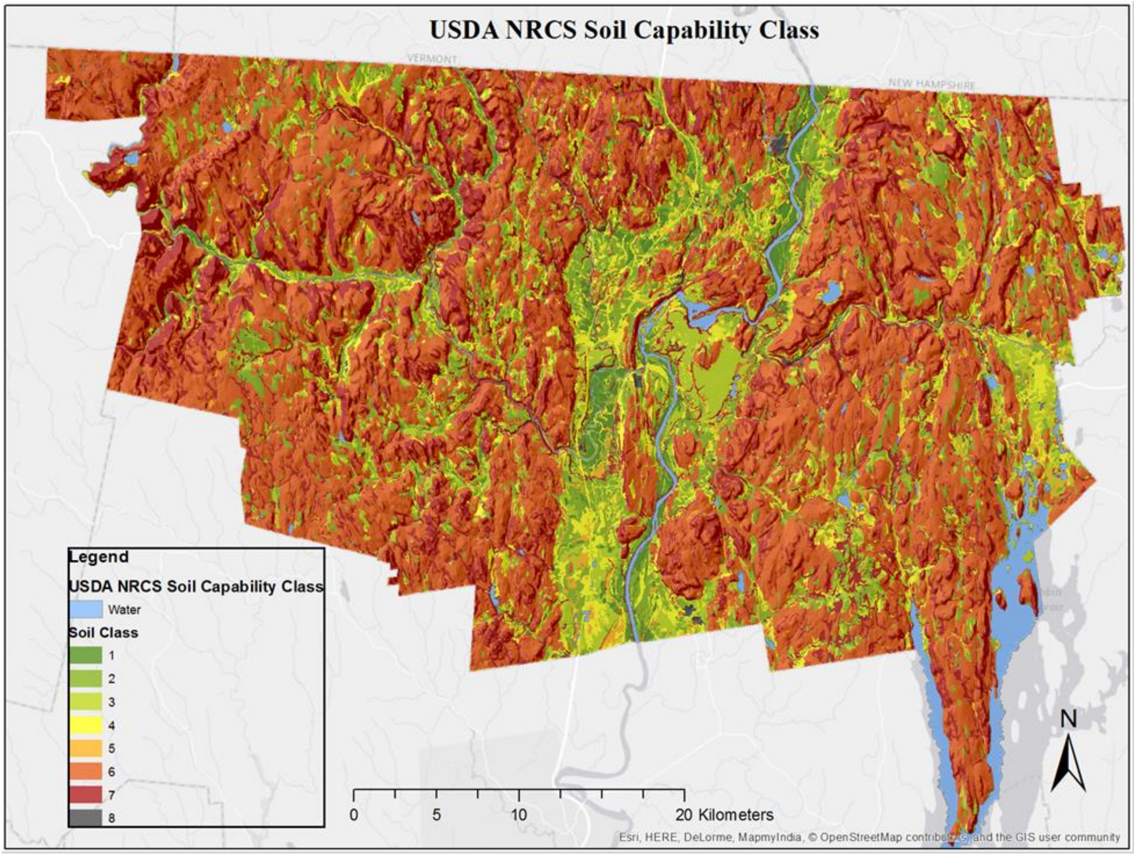


Figure 7. USDA NRCS soil capability class spatial distribution. Note the limited amount of prime farmland illustrated in green and its concentration in the Connecticut River Valley.

Scenario Design

In order to achieve the main objective of identifying and analyzing the tradeoffs from different potential land-use changes, I created five unique scenarios in addition to the 2005 baseline. All five scenarios were designed to deliberately test several assumptions outlined in the NEFV policy goals, chiefly if the landscape could dramatically increase local food production, and to examine the ecosystem service consequences. Each scenario is not intended to be a complete prediction of future circumstances, but to illustrate particular land-use patterns capable of informing public policy decisions and their consequences.

The first four scenarios were designed to model the ecosystem service tradeoffs associated with possible agriculture expansion regimes in the next 50 years. Additionally, a retrospective fifth scenario was created to illustrate the 1830s land use impacts of wide scale deforestation with low intensity agricultural activity that consisted exclusively of grazing livestock. The major underlying assumption across all five scenarios is that agriculture will expand from its current base in 2005.

The first four scenarios were created by making assumptions as to how agriculture might expand based on the availability and quality of land in Franklin County. Further assumptions were made based on the USDA soil capability class ranking system. Because current agriculture already exists on 15,079 hectares (8.3%), it was initially excluded from the process of identifying land areas suitable for future expansion. Additionally, I excluded all land in land-use classes designated as developed or that we deemed not suitable for agriculture, including “low-density residential” and “forested-wetlands”. We only expanded agriculture activity onto the land-use classes of forest, openland, and brushland / successional. Nearly all of the land area available for agriculture expansion occurs in forested areas. This reflects the fact that 76.8% of the county is forested and that expansion onto other land-use classes would not be realistic for future farming activities. We also decided not to include mixed land-use classes for any given pixel as this would present significant modeling challenges with the InVEST software. Mixed land uses are often a reality for agriculture expansion and backyard gardening in low density residential areas can produce a significant amount of food, but we decided to exclude these types of land uses from the model and to limit the scale of a farm to one pixel size of $30m^2$ and above.

The fifth scenario is based on a reconstruction of the agricultural extent present in the 1830s, using a map of known forested areas (Foster, 2006). This initial 1830s data layer proposed modeling challenges because a significant portion (26%) of the county had missing data. We recreated the land-use mix for the missing areas using a series of GIS processes that calculated the likelihood of forest cover based on slope and distance to know roads.

2005 Baseline: Current Land-use Pattern

The baseline scenario is considered to be the current land-use pattern and the one with the highest degree of conservation due to the unique land-use history of the region. This scenario was calculated directly from the MassGIS Land Use 2005 data layer (MassGIS, 2015). According to the land use classifications by MassGIS, the 2005 baseline land pattern indicates a landscape that is predominately forested at 76.8% (144,177 ha). Row crop cultivation is largely concentrated in the Connecticut River Valley and accounts for 5.3% (9,612 ha). Pasture land is dispersed in relatively small parcels throughout the county, totaling 2.8% (4,987 ha). Non-forested wetlands account for 1.9% (3,477 ha). Low-Density Residential accounts for 1.8% (3374 ha). Forested Wetlands account for 1.8% (3293 ha). Very Low Density Residential accounts for 1.4% (2,711 ha). Orchard areas are less significant at 0.3% (480 ha) of the total land area.

Water bodies account for 3.5% (6,653 ha) of the county area. Figure 8 shows the geospatial distribution of the 2005 land-use pattern.

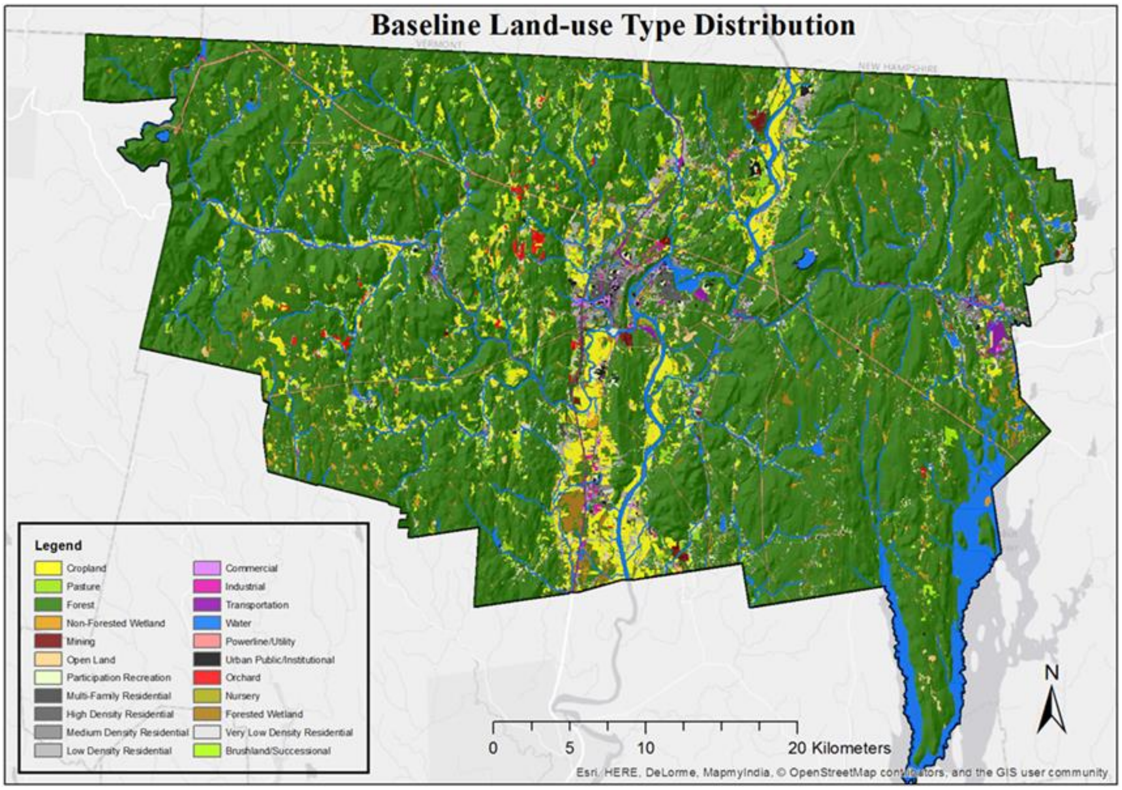


Figure 8. Map of baseline land-use type distribution. Note the predominance of forested land cover.

Scenario 1: Maximum Agriculture Expansion with No Restrictions

This scenario assumes widespread agriculture expansion with the intent of exploring the maximum agriculture capacity of the county and its associated impacts. With this scenario we seek to understand the implications from maximum agriculture output and to explore the feasibility of regional self-sufficiency. Under this scenario most of the marginal and subprime forested land is cleared and converted pasture. Land that is currently pasture is converted to crop production. This scenario envisions drastically higher energy costs with market forces that discourage importing food. Food and fuel will be at a premium as the region struggles to cope with energy shortages. Forest conservation is a low priority and only areas that are currently difficult to access are not cleared. Under this scenario the landscape will resemble the region circa 1850, which was the peak of deforestation for agriculture production, but will include a significant expansion of cultivated cropland.

After spatially identifying the pool of available land area suitable for agriculture expansion by land use class, the NRCS SSURGO-Certified Soils datalayer was used to identify areas for conversion to either pasture or cropland. All soil classifications including those with moderate (classes 3 and 4) and very severe limitations (classes 5 and 6) are included. Soil classes 1, 2, and 3 are converted into cultivated cropland. Soil classes 4, 5, and 6 are converted into pastureland. Only soil classes 7 and 8 are excluded from agriculture expansion. Conservation interests are not considered under this scenario. Development does not increase beyond the baseline. Conversion only occurs on land cover types of forest, openland, and brushland / successional. Under this scenario cropland increased to 16% (29,891 ha) of the county land area; pasture increased to 49% (91,776 ha), and forest was reduced to 22% (40,780 ha). Figure 9 shows the geospatial distribution of the scenario 1 land-use pattern.

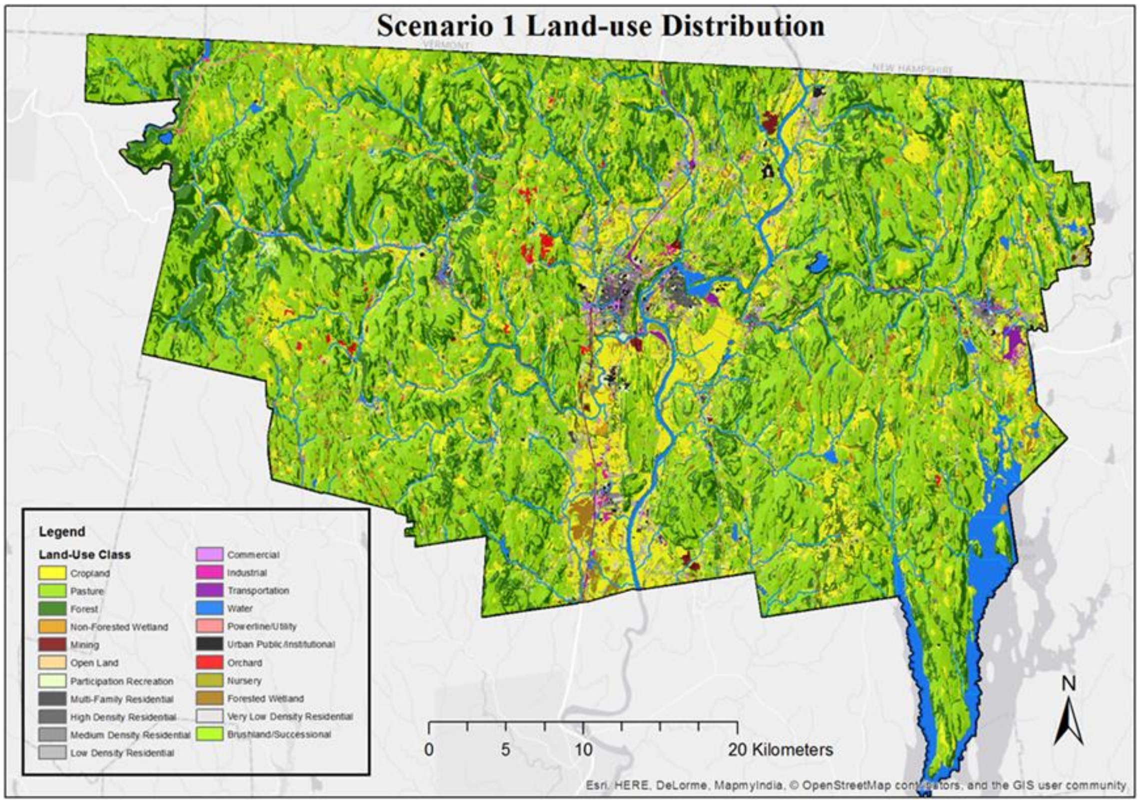


Figure 9. Map of scenario 1 land-use type distribution. Note the predominance of pastureland and the expansion of cropland into currently forested land cover.

Scenario 2: Agriculture Expansion with Current Conservation

The second scenario was created using a continuation of procedures described in Scenario 1, but incorporates existing conservation structures. Land areas with permanent or temporary conservation restrictions were removed from the identified pool of convertible land. Temporary conservation interests included parcels managed to maintain current use under the Massachusetts chapter 61, 61A, 61 B programs, which place restrictions on enrolled parcels in 10 year (ch. 61) and 5 year (ch. 61A, 61B) increments. The chapter 61 parcels were identified from the Mass GIS Stewardship Program Properties data set, clipped to Franklin County and removed from the pool of expandable land areas.

Long term conservation restrictions were also excluded from agriculture expansion. We defined long term conservation as land with legal structures restricting land use in perpetuity. These lands are often owned by private land owners but their development rights have been restricted for conservation purposes by the State, Town or non-profit organizations. These lands were identified using the MassGIS Protected and Recreational OpenSpace data set. The various types of conservation classifications and structures, including Conservation Restricted (CR) and Agriculture Preservation Restriction (APR) were consolidated into a single shapefile and removed from the pool of expandable land areas. Under this scenario cropland increased from the baseline to 12% (22,291 ha) of the county land area; pasture increased to 27% (51,530 ha), and forest was reduced to 47% (87,551 ha). Figure 10 shows the geospatial distribution of the scenario 2 land-use pattern.

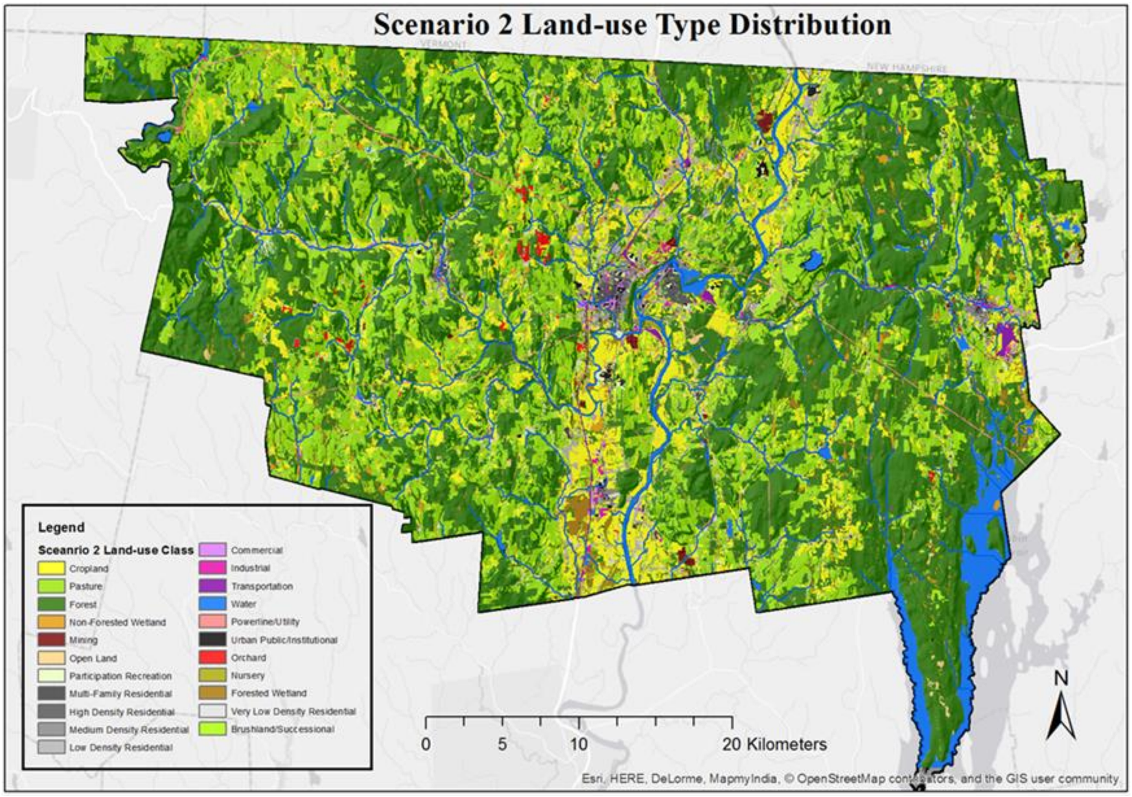


Figure 10. Map of scenario 2 land-use type distribution. Note the distribution of present day conserved forested land cover.

Scenario 3: Agriculture Expansion with Current and Future Conservation

The third scenario builds upon the second scenario by incorporating existing conservation plans as well as future conservation goals. Just as in the first two scenarios soil classes converted into cultivated cropland include classes 1, 2 and 3 and soil classes 4, 5 and 6 are converted into pastureland. As described in Scenario 2, land areas with temporary and permeant conservation easements were removed from agriculture expansion.

The MassGIS BioMap2 data set is designed to guide strategic efforts for future conservation of biodiversity. The BioMap2 data set spatially identifies areas critical to endangered species and rare natural environments. We combined two components of the

BioMap2 data set including Core Habitat and Critical Natural Landscape to create a shapefile that spatially identifies areas critical to future conservation goals. The Core Habitat (CH) layer spatially identifies rare natural communities and intact ecosystems necessary to promote species listed under the Massachusetts Endangered Species Act in addition to species listed in the State Wildlife Plan. The Critical Natural Landscape (CNL) layer identifies intact and unfragmented natural landscapes most capable of supporting ecological processes including disturbance regimes, species diversity, and the preservation of a wide range of habitats. We combined the CH and CNL layers and removed these areas from the pool of convertible land. Under this scenario cropland increased from the baseline to 9% (16,791 ha) of the county land area; pasture increased to 14% (25,509 ha), and forest was reduced to 62% (117,249 ha). Figure 11 shows the geospatial distribution of the scenario 3 land-use pattern.

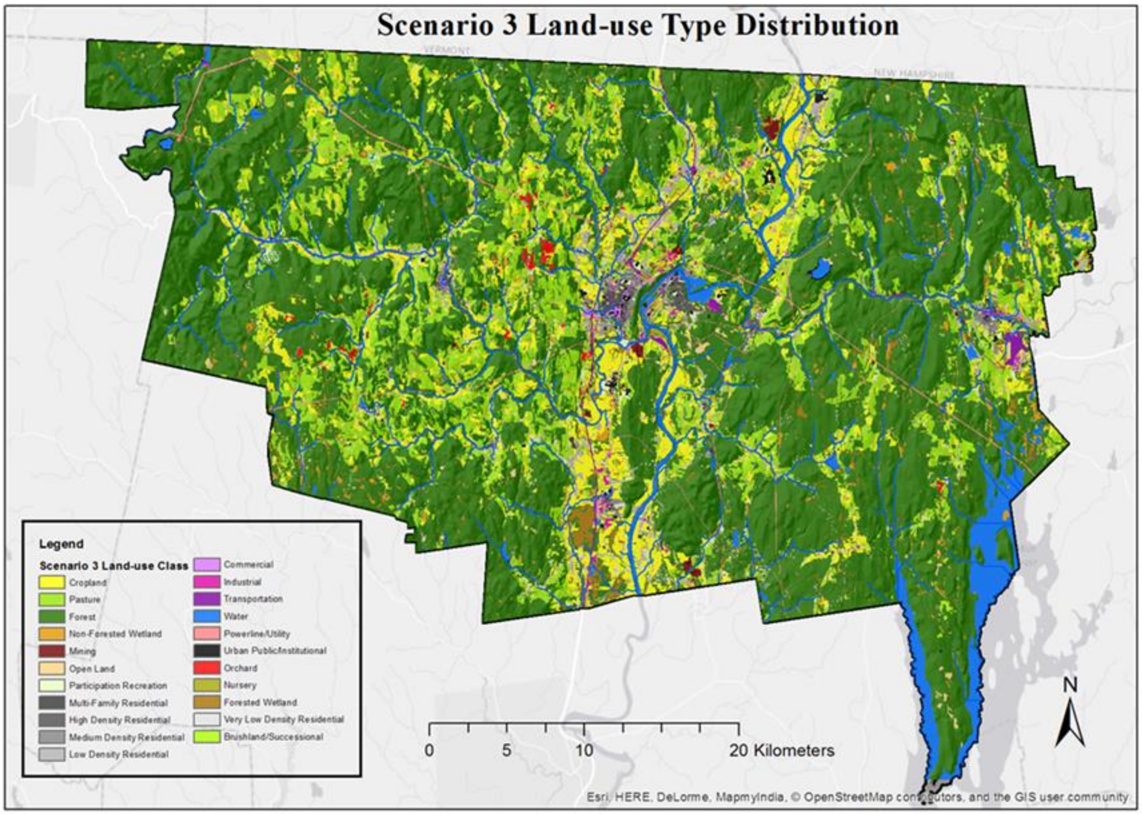


Figure 11. Map of scenario 3 land-use type distribution. Note the relative proportion of forested land cover as a result of current and future conservation plans.

Scenario 4: Continuous Expansion with Current and Future Conservation

The purpose of this scenario is to examine the realistic feasibility of the NEFV on a landscape scale. Under this scenario roughly 70% of the landscape will remain as forested, with agriculture maximized on the remaining areas. Growth of the built environment remains at 2005 baseline levels. Agriculture production type will be determined by soil capability class and NEFV dietary ratios. The feasibility of this scenario will be measured against the backdrop of the other scenarios to examine its shortcomings in achievable agriculture production and sacrifices in ecosystem services. This scenario attempts to project a modest increase in food production while minimizing the sacrifices to ecosystem services.

The fourth scenario is a continuation of scenario 3, but is more restrictive because it assumes that future agriculture expansion will occur immediately adjacent to land areas with current agriculture. The logic for this scenario assumes that farmers will be the vanguard of agriculture expansion and that they will be the agents of change. This implies that farmers will attempt to expand their operations based on their knowledge of current production systems and local proximity to current infrastructure.

There are two major assumptions that direct the progression of agriculture expansion under this scenario. The first assumes that agriculture expansion will occur within close proximity (283 meters, or 75th quantile) to current pasture or cropland. This has the effect of removing outliers and projecting isolated pockets of agriculture expansion deep within the forest boundary.

The second decision rule assumes that existing pasture will be converted into cultivated cropland if the soil capability class is no greater than 3, and existing forest will be converted to new pasture if the soil capability class is no greater than 5. Unlike the first three scenarios, where only soil classes 7 and 8 were excluded from the pool of potential expansion, in Scenario 4 we chose to exclude soil classes 6, 7 and 8 from agriculture expansion. This decision was made on the premise that soil class 6 is best suited as forest because of its marginal to poor site attributes, whereas we chose to convert soil class 6 into pasture in the first three scenarios. The exclusion of soil class 6 is significant because it amounts to nearly half (47%) the land area in the county. The intention of these decision-rules for this scenario is to design a more sustainable option. Under this scenario cropland remained constant at 5% of the county land area, but decreased slightly by 140 hectares from the baseline scenario to 9,473 hectares. Pasture increased to 7% (13,423 ha), and forest was reduced to 73% (136,791 ha) of the county. Figure 12 shows the geospatial distribution of the scenario 4 land-use pattern.

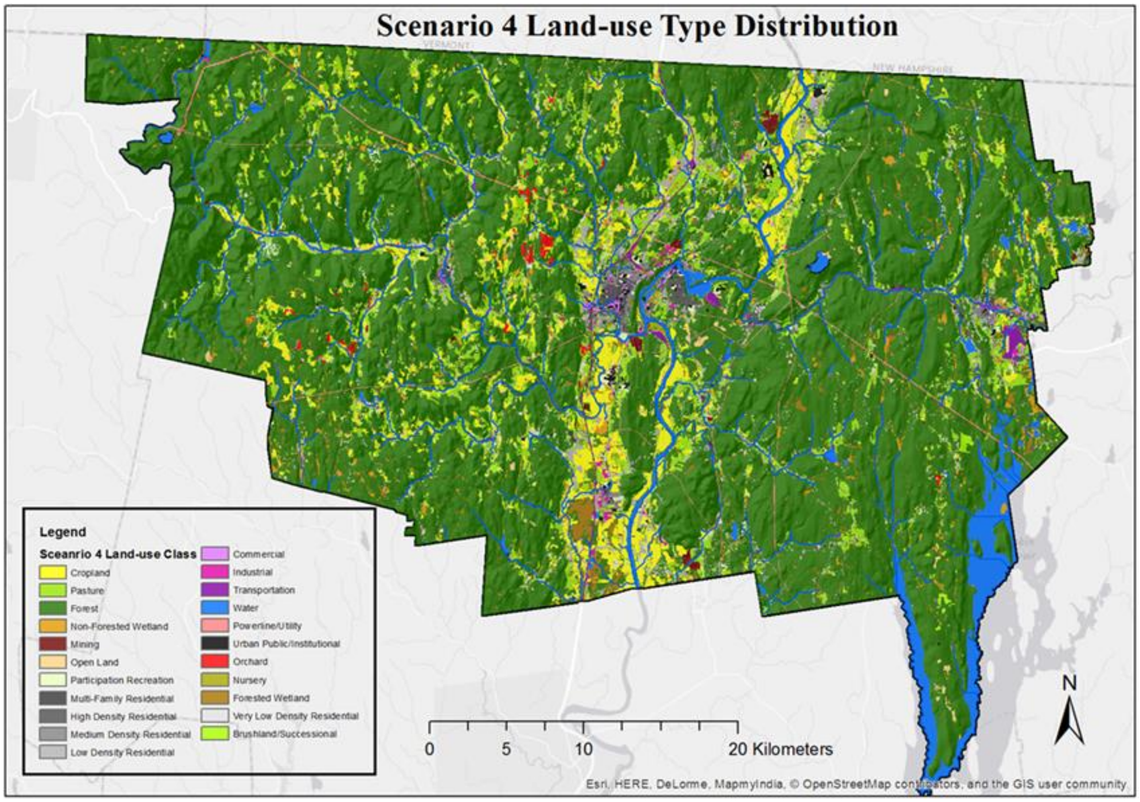


Figure 12. Map of scenario 4 land-use type distribution. Note the relative proportion of forested land cover as a result of current and future conservation plans in addition to built-in mechanism to limit agriculture expansion to occur adjacent current infrastructure.

Scenario 5: Pastoral Expansion (1830s land use pattern)

The fifth scenario attempts to model the ecosystem impact from wide scale agriculture expansion and deforestation leading to a predominantly pastoral landscape, by examining the effect of returning to the 1830s in the year 2060. This scenario was created as an academic exploration of land-use changes and associated impacts on ecosystem services from a historical perspective. As noted in published literature (Foster, 2006; D.

R. Foster et al., 2010), massive landscape transformations have occurred in Franklin County across the past three centuries.

To simulate the peak of deforestation I used a land-use map for the1830s created by the Harvard Forest research group (Hall, 2002). The 1830s land-use map was created by compiling local records on known forest parcel locations, size and type, and then used as the basis for creating the fourth scenario. Although a significant portion of the county land cover data has been compiled, seven out of twenty six towns or roughly 26% of the land area in northwestern corner the county had inadequate land cover data to reconstruct a complete map as seen in Figure 13.

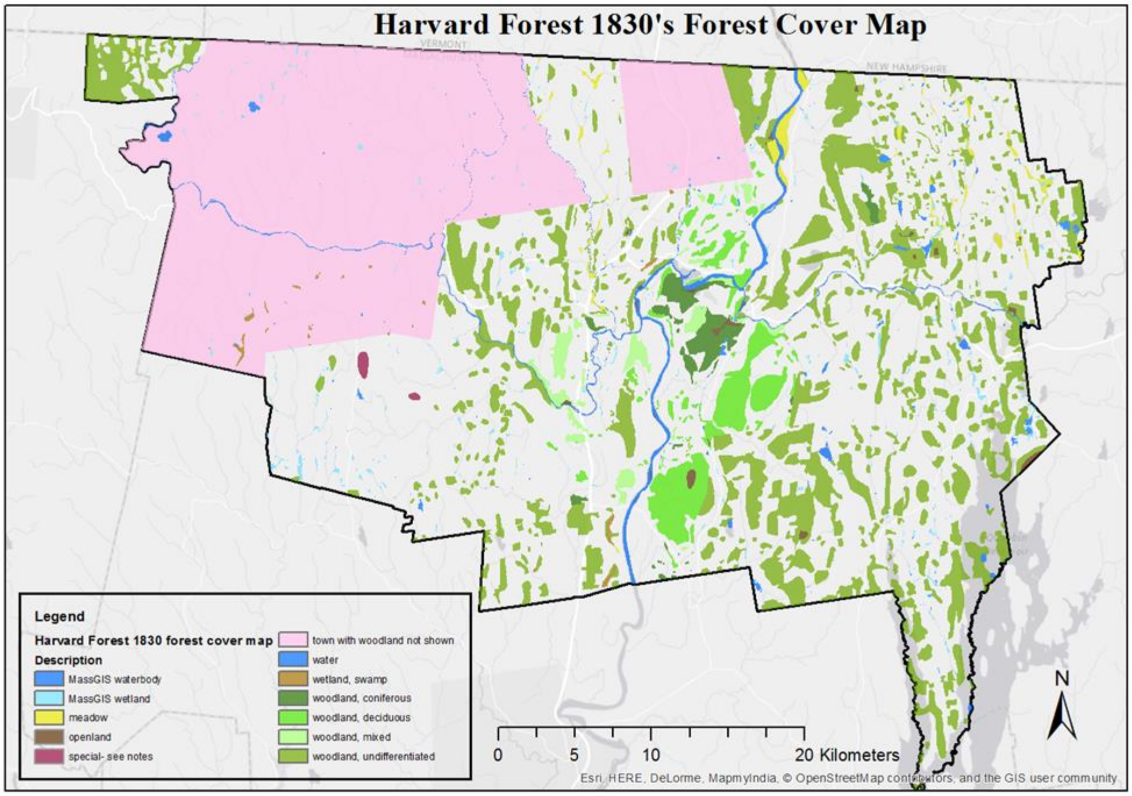


Figure 13. Harvard Forest's 1830s forest cover data. Note the geospatial distribution of known woodlots and the large proportion of the county with missing data shown in pink. The majority of land cover during this period was deforested.

To reconstruct the missing land-cover data I identified two major variables affecting land transformation during the 1830s, both revolving around the farmer's ability to clear forest and accessibility. These variables of slope and distance to existing roads were the primary factors driving deforestation. In the towns with missing data, land areas with a slope greater than 20% and a distance greater than 570 meters (75th quantile) from existing roads were assumed to be forested.

In order for the InVEST module to function properly each pixel on the land use raster must be assigned a land-use classification. The remaining land area that was not forested or with no identified land classification data was assumed to be pasture. The source data from Harvard Forest implies that these areas were most likely "agriculture in nature or open pastures" (Hall, 2002). We chose to assign these non-forested areas as "pasture" to not confound the nutrient retention model because contemporary agriculture methods and rural land management during the 1830s differed significantly from the nutrient export coefficients of 20th century that were used to create the biophysical table in InVEST.

As a decision rule for the fifth scenario we chose to exclude the cultivated crop land-use class. Current understanding of nutrient export coefficients for the time period do not adequately depict impact of cultivated crops circa the 1830s. For this reason we chose to completely exclude cropland from the scenario and to project all areas non-forested (no data raster cells) as pasture. Pasture was deemed a reasonable classification because of the moderate range of biophysical properties that mimicked the general land use characterized by grazing and low intensity crop production during the 1830s. Because of the lack of adequate biophysical data for the time period, the domain of inference is limited to the expansion of pasture from forest. Under this scenario pasture increased to 70% (131,296 ha) above the baseline, and forest was reduced to 27% (51,452 ha) of the county. Figure 14 shows the geospatial distribution of the scenario 5 land-use pattern.

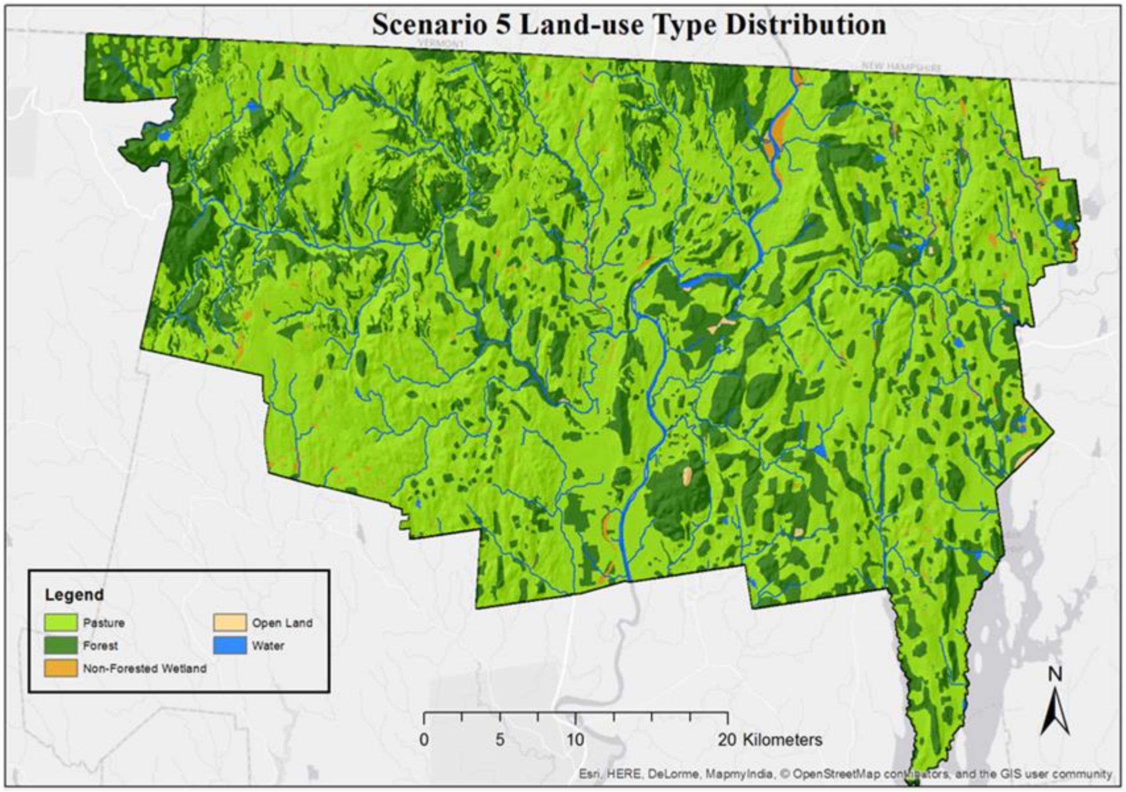


Figure 14. Map of scenario 5 land-use type distribution. Note the distribution of forested land cover and the predominance of pastureland. Also note the reconstruction of missing forest data with estimations based on slope and distance to known roads. This scenario depicts the four known land use types for the 1830s map.

Figure 15 shows the flow chart of GIS procedures for creating each scenario. Figure 16 shows the percentage of land reallocated into each land-use class, and the relative proportion of forest land converted into pasture or cropland from forest.

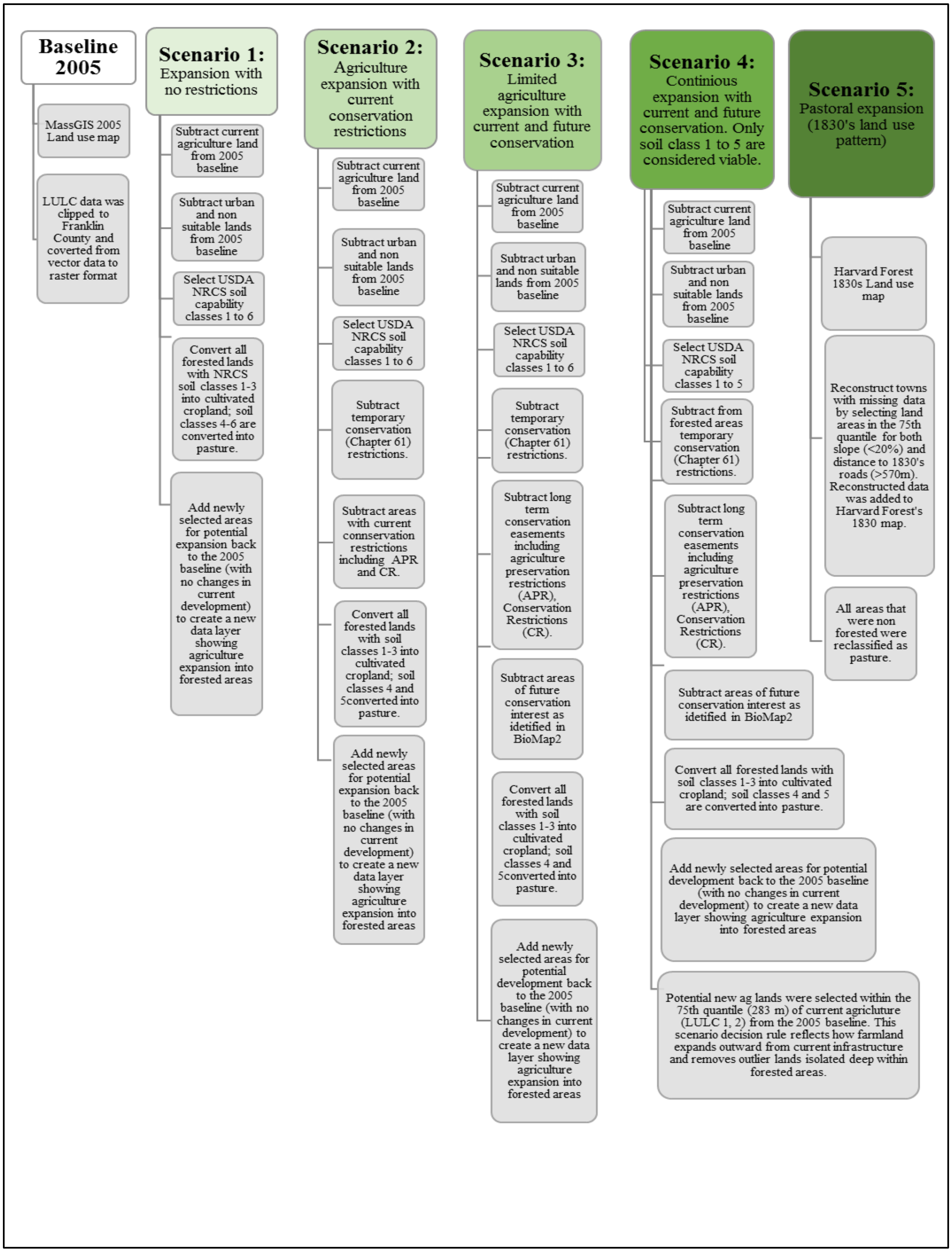


Figure 15. Flow chart showing the GIS procedures for each scenario.

Scenario profiles and the relative land-use composition has been summarized into Table 2, which shows the total area allocated to each land-use type based on the decision rules outlined in the methods. Figures 16 and 17 show the relative proportion of forest, crop and pasture land in each scenario.

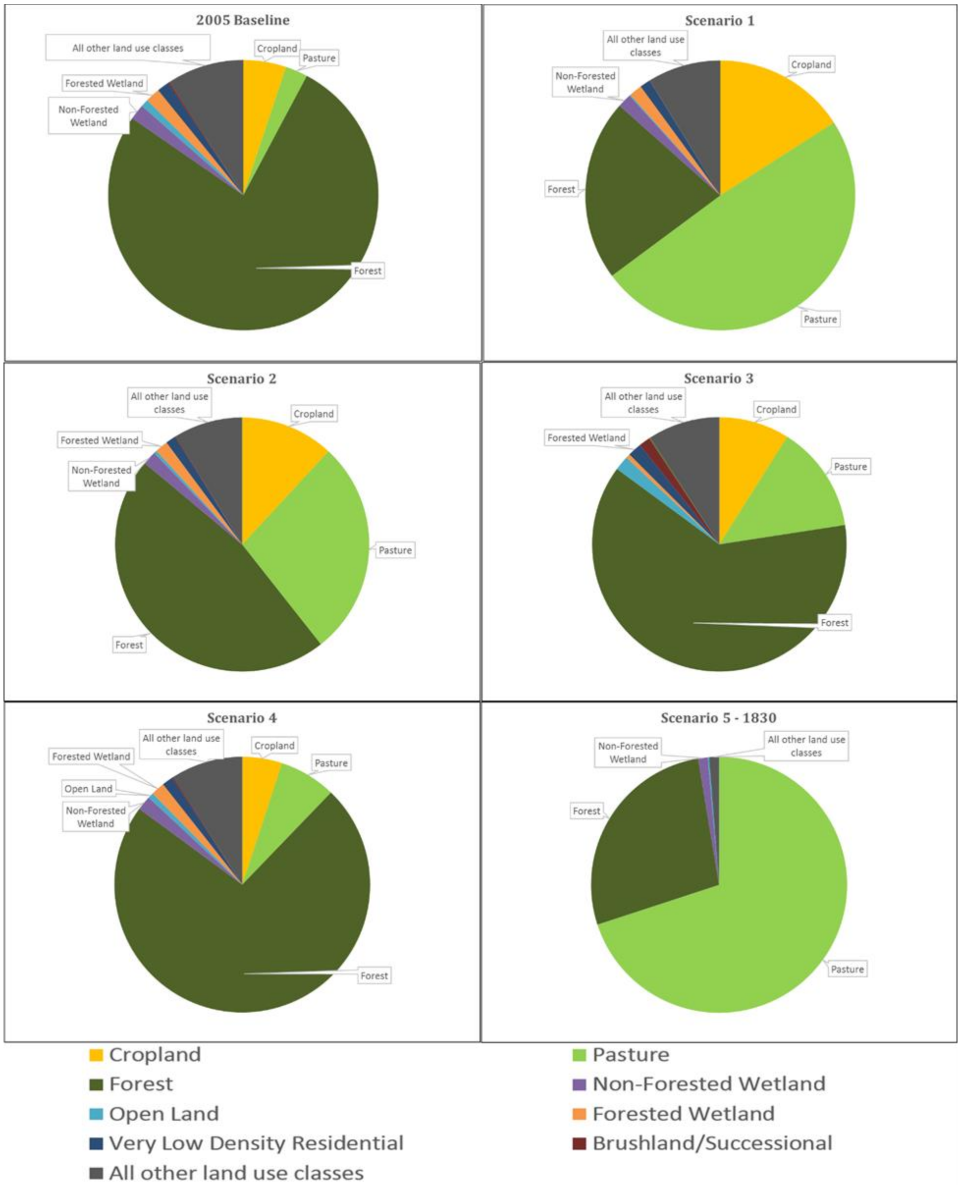


Figure 16. Pie charts of relative percentage of land-use classes under each scenario. Note the build-out of cropland in scenario 1, as well as the absence of cropland under scenario 5. Future conservation goals are reflected in scenarios 3 and 5.

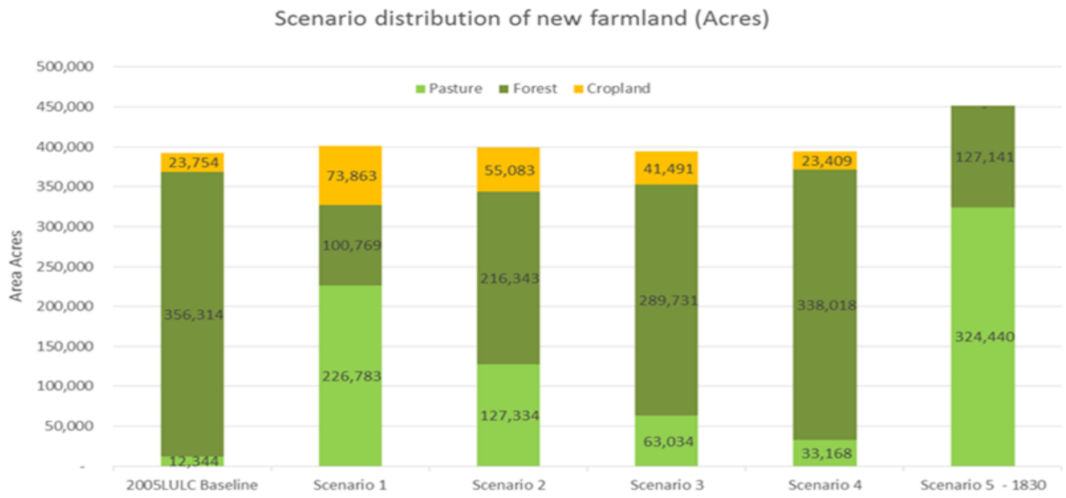


Figure 17. Relative conversions of forested land cover into cropland and pasture. Note that scenario 5 has more total land area than the other scenarios because the Quabbin reservoir was not built at that time and the land area now submerged was actively farmed.

Table 3. Scenario composition by land-use class.

	2005 Baseline		**Scenario 1**		**Scenario 2**		**Scenario 3**		**Scenario 4**		**Scenario 5 - 1830**	
LULC Name	Hectares	% of total Area	Hectares	% of total Area	Hectares	% of total Area	Hectares	% of total Area	Hectares	% of total Area	Hectares	% of total Area
Cropland	9,613	5%	29,891	16%	22,291	12%	16,791	9%	9,473	5%	-	0%
Pasture	4,995	3%	91,776	49%	51,530	27%	25,509	14%	13,423	7%	131,296	70%
Forest	144,195	77%	40,780	22%	87,551	47%	117,250	62%	136,791	73%	51,452	27%
Non-Forested Wetland	3,473	2%	3,148	2%	3,278	2%	3,473	2%	3,410	2%	2,264	1%
Open Land	1,711	1%	262	0%	719	0%	1,036	1%	1,344	1%	314	0%
Forested Wetland	3,294	2%	2,925	2%	3,075	2%	3,294	2%	3,240	2%	-	0%
Very Low Density Residential	2,704	1%	2,387	1%	2,401	1%	2,704	1%	2,610	1%	-	0%
Brushland/Successional	360	0%	59	0%	228	0%	276	0%	315	0%	-	0%
All other land use classes	17,297	9%	16,413	9%	16,423	9%	17,158	9%	17,037	9%	2,339	1%

Total ha of each land-use type and the % total land area within Franklin County.

Nutrient Retention Modeling

Once the scenarios were mapped they were used to model the impacts on ecosystem services using the InVEST software package. The Nutrient Retention module version 3.1.0 was used to compute the nutrient runoff for both nitrogen and phosphorus for each of the 228 sub watersheds contained within or immediately adjacent to the Franklin County boundary. The Nutrient Retention module required several data inputs and reference tables to compute the nutrient load, absorption and runoff from each subwatershed in Franklin County. Several of the required module inputs could be downloaded from the MassGIS data library (MassGIS, 2015), including the: Digital Elevation Model (DEM 1:5,000) (2005), Crop Evapotranspiration and Potential Evaporation Grids (2005), Land Use (2005), Drainage Sub-basins (2007), (MassGIS, 2015). The soil depth to root restricting layer and plant available water fraction attributes were obtained from the NRCS SSURGO-Certified Soils datalayer for Franklin County. Serval additional data requirements were adapted from published research, including the biophysical table, and water purification threshold table (Blumstein & Thompson, 2015; Lin, 2004; Reckhow, Beaulac, & Simpson, 1980). The biophysical inputs remained constant for each of the scenarios except for the distribution of land-use classes as defined by each scenario. Figure 18 shows the spatial orientation of the five watershed and the portion contained within the county boundary.

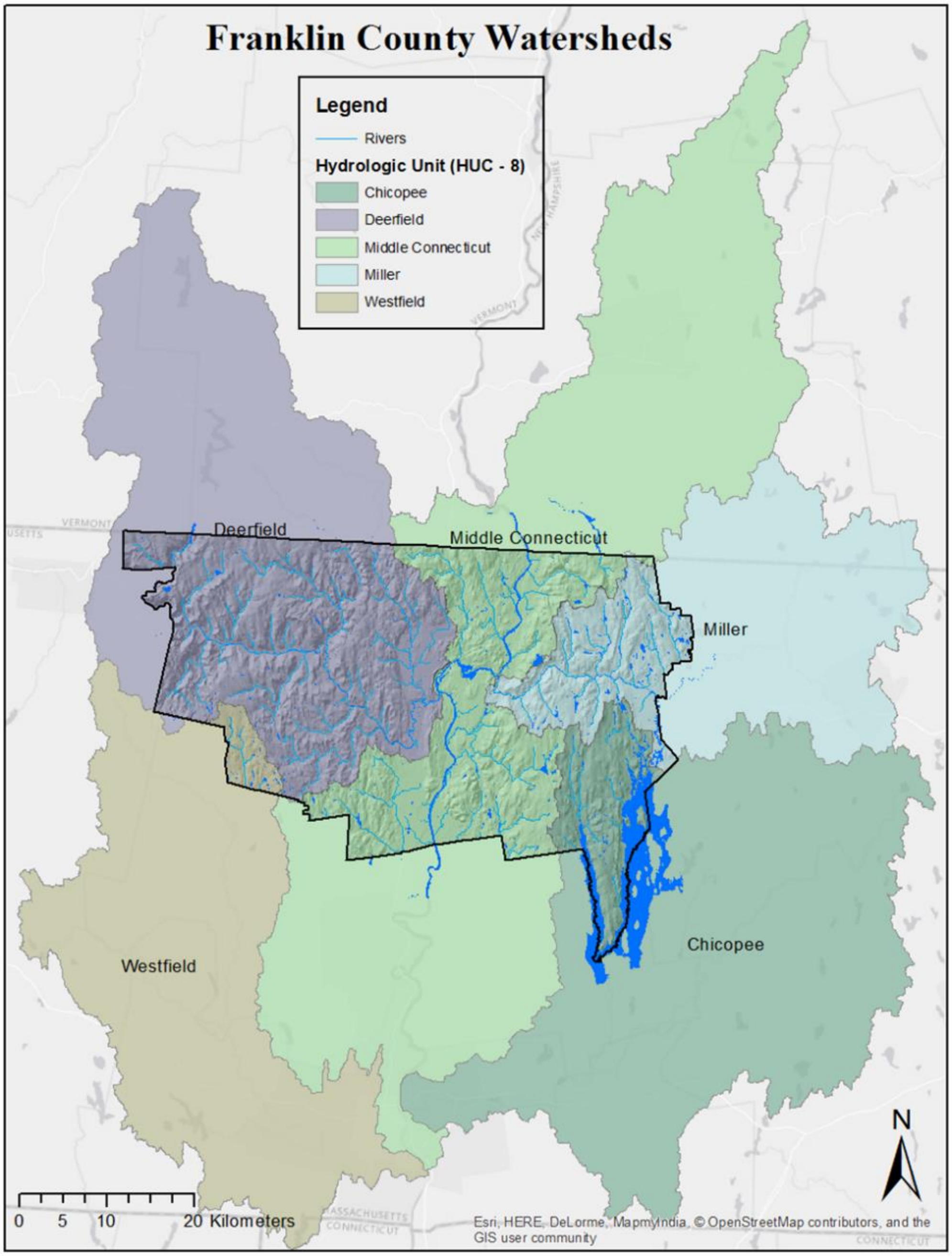
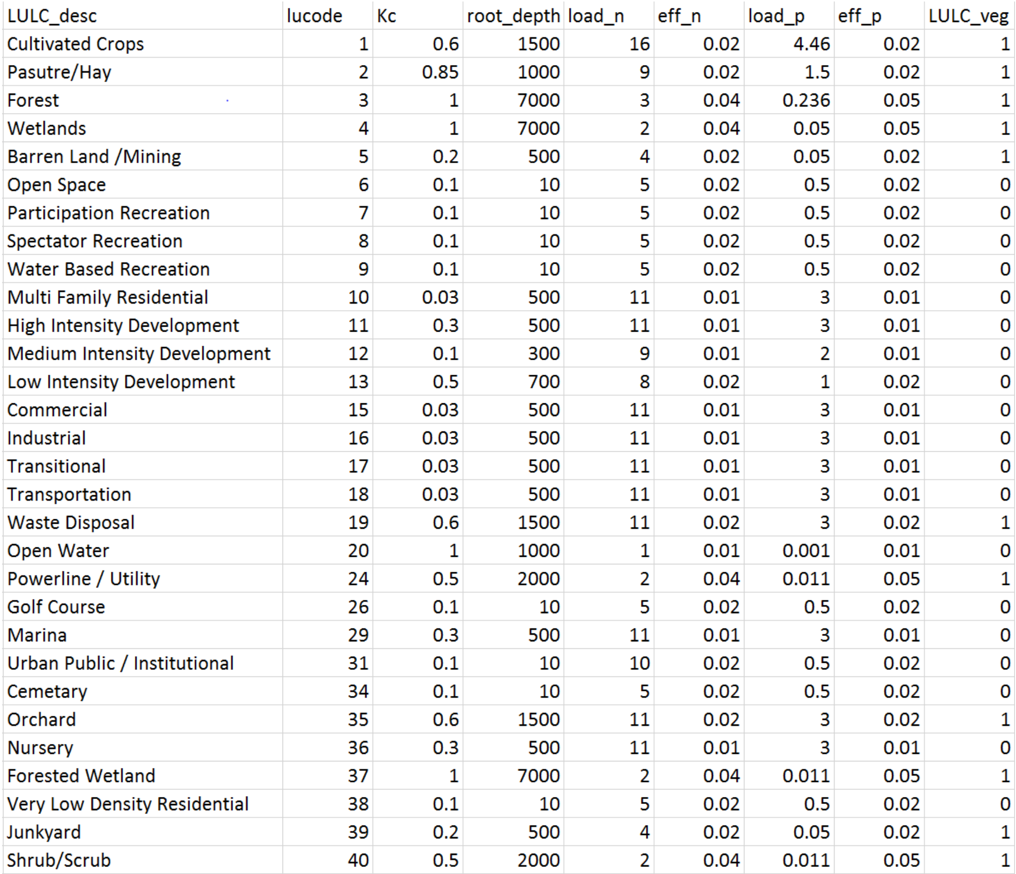


Figure 18. Map of major watersheds and the Franklin county boundary.

The biophysical table was constructed from water quality coefficients assigned to each land use class as seen in Table 4. Additional attributes for of the biophysical table included the land use code (lucode), land class description (LULC_desc), maximum root depth in millimeters (root_depth), vegetative evapotranspiration coefficient (Kc), nitrogen nutrient load (load_n / load_P), and natural vegetation filtering capacity (eff_n / eff_p). The threshold flow accumulation value was set to 1,000 which visually matched the geospatial stream network for the county.

Table 4. InVEST nutrient retention model input data.



LULC_desc	lucode	Kc	root_depth	load_n	eff_n	load_p	eff_p	LULC_veg
Cultivated Crops	1	0.6	1500	16	0.02	4.46	0.02	1
Pasutre/Hay	2	0.85	1000	9	0.02	1.5	0.02	1
Forest	3	1	7000	3	0.04	0.236	0.05	1
Wetlands	4	1	7000	2	0.04	0.05	0.05	1
Barren Land /Mining	5	0.2	500	4	0.02	0.05	0.02	1
Open Space	6	0.1	10	5	0.02	0.5	0.02	0
Participation Recreation	7	0.1	10	5	0.02	0.5	0.02	0
Spectator Recreation	8	0.1	10	5	0.02	0.5	0.02	0
Water Based Recreation	9	0.1	10	5	0.02	0.5	0.02	0
Multi Family Residential	10	0.03	500	11	0.01	3	0.01	0
High Intensity Development	11	0.3	500	11	0.01	3	0.01	0
Medium Intensity Development	12	0.1	300	9	0.01	2	0.01	0
Low Intensity Development	13	0.5	700	8	0.02	1	0.02	0
Commercial	15	0.03	500	11	0.01	3	0.01	0
Industrial	16	0.03	500	11	0.01	3	0.01	0
Transitional	17	0.03	500	11	0.01	3	0.01	0
Transportation	18	0.03	500	11	0.01	3	0.01	0
Waste Disposal	19	0.6	1500	11	0.02	3	0.02	1
Open Water	20	1	1000	1	0.01	0.001	0.01	0
Powerline / Utility	24	0.5	2000	2	0.04	0.011	0.05	1
Golf Course	26	0.1	10	5	0.02	0.5	0.02	0
Marina	29	0.3	500	11	0.01	3	0.01	0
Urban Public / Institutional	31	0.1	10	10	0.02	0.5	0.02	0
Cemetary	34	0.1	10	5	0.02	0.5	0.02	0
Orchard	35	0.6	1500	11	0.02	3	0.02	1
Nursery	36	0.3	500	11	0.01	3	0.01	0
Forested Wetland	37	1	7000	2	0.04	0.011	0.05	1
Very Low Density Residential	38	0.1	10	5	0.02	0.5	0.02	0
Junkyard	39	0.2	500	4	0.02	0.05	0.02	1
Shrub/Scrub	40	0.5	2000	2	0.04	0.011	0.05	1

Biophysical inputs adapted from (Blumstein & Thompson, 2015; Reckhow et al., 1980).

The nutrient retention model works by computing a load value for each pixel based assigned nutrient coefficients for each land use class. The model does not account for point source pollution, but assumes non-point source pollution from land-use changes. The annual precipitation and DEM layers were used to calculate the flow rate for each pixel assuming water flows down elevation gradients. The filtration rate is subtracted from the load to produce the retention capacity and net export for each pixel until reaching a stream or river. Significant biophysical changes in land use class or vegetation cover can affect the nutrient loading capacity, but flow rate and absorption rate have the ability to reduce overall export because the pixel algorithm synergistically accumulates nutrient concentration downstream (Sharp, 2015). Valuation of pixel export is affected by this behavior because the load accumulation downstream from non-point sources is most likely where the pollutants have the greatest eutrophic impact in rivers, ponds, lakes and estuaries.

The valuation of nutrient exports was calculated from the cost to remove one kilogram of nutrient pollutant from the watershed. There is poor data on the costs associated with non-point source pollution treatment with decentralized treatment facilities (EPA, 2015; Houle, Roseen, Ballestero, Puls, & Sherrard Jr, 2013). I chose to use published estimates from the Chesapeake Bay assessment of nutrient abatement projects (Commission, 2004). The report outlines several options for reducing the impact from agriculture induced non-point source nutrient pollution into the Chesapeake Bay, and I chose to run the nutrient retention model with two different nutrient remediation strategies to establish upper and lower cost estimates. The social cost estimates were based off of the Chesapeake Bay financial assessment for non-point source nutrient loading from heavy agriculture activity. I modeled both the higher costs of treatment associated with Waste Water Treatment (WWT) plants, as well as implementing on-farm best management practices that include farm specific Nutrient Management Plans (NMP). The Chesapeake Bay cost estimates for annual nitrogen and phosphorus remediation with WWT were estimated at $3.88 and $33.57 per kg/watershed, respectively. The annual cost of implementing the NMP for nitrogen was estimated at $0.75 per kg/watershed and $12.82 for phosphorus. The market discount rate was estimated at 3% for the 50 year time frame.

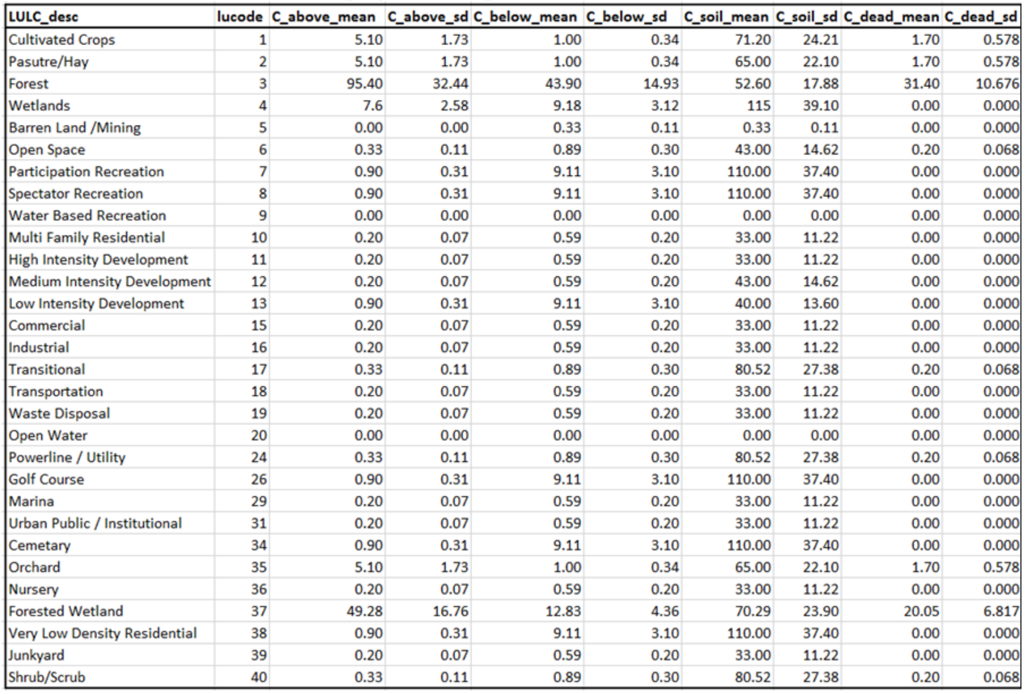
Estimating the social cost from additional nutrient runoff from non-point sources requires a complex analysis that is beyond the scope of this project. Several variables needed to calculate the social costs estimates were not available at the time of this analysis, including the Total Maximum Daily Load (TMDL) for the 5 watersheds included in this study. I also did not take into account the specific WWT capacity and associated costs from improving these estimates. I would also have had to develop a far more comprehensive model capable of replicating the impacts of biological processes affecting nutrient cycling. The InVEST Nutrient Retention model 3.1.1 used in this study is not capable of estimating these complexities. For these reasons we chose to adopt the Chesapeake Bay estimates because they reflect a landscape with a heavy agriculture footprint and its correlated nutrient export and the costs to society. The costs of nutrient export in this analysis are not intended be accurate appraisals for Franklin County, but rather a relative comparison of the social costs from forgone ecosystem services.

The model outputs were aggregated into summary tables for each of the scenarios. The results were arranged into four main categories for both Nitrogen and Phosphorous including: total load of (P or N) available per watershed (kg/ha), total amount of (P or N) retained by the landscape (kg/watershed), total amount of (P or N) exported into the watershed (kg/watershed, and the total value of (P or N) exported to the stream in the watershed (USD/kg).

Carbon Storage Modeling

The InVEST Carbon Storage and Sequestration model works by computing the volume of carbon in each cell of raster map created for each scenario. Every cell in the raster maps was assigned a land use class and each land use class was assigned carbon storage coefficients. There are four pools of carbon for each land use type including: aboveground biomass, belowground biomass, soil, and dead organic matter. The rates of carbon storage were compiled into a single biophysical table listing all of the land use classes and the assigned an estimate of the carbon coefficients for each of the four pools (see Table 5). The model aggregates and estimates the net amount of carbon stored in each pixel on the land use map according to the biophysical input data.

Table 5. Carbon model biophysical input variables for each land-use class.



LULC_desc	lucode	C_above_mean	C_above_sd	C_below_mean	C_below_sd	C_soil_mean	C_soil_sd	C_dead_mean	C_dead_sd
Cultivated Crops	1	5.10	1.73	1.00	0.34	71.20	24.21	1.70	0.578
Pasutre/Hay	2	5.10	1.73	1.00	0.34	65.00	22.10	1.70	0.578
Forest	3	95.40	32.44	43.90	14.93	52.60	17.88	31.40	10.676
Wetlands	4	7.6	2.58	9.18	3.12	115	39.10	0.00	0.000
Barren Land /Mining	5	0.00	0.00	0.33	0.11	0.33	0.11	0.00	0.000
Open Space	6	0.33	0.11	0.89	0.30	43.00	14.62	0.20	0.068
Participation Recreation	7	0.90	0.31	9.11	3.10	110.00	37.40	0.00	0.000
Spectator Recreation	8	0.90	0.31	9.11	3.10	110.00	37.40	0.00	0.000
Water Based Recreation	9	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.000
Multi Family Residential	10	0.20	0.07	0.59	0.20	33.00	11.22	0.00	0.000
High Intensity Development	11	0.20	0.07	0.59	0.20	33.00	11.22	0.00	0.000
Medium Intensity Development	12	0.20	0.07	0.59	0.20	43.00	14.62	0.00	0.000
Low Intensity Development	13	0.90	0.31	9.11	3.10	40.00	13.60	0.00	0.000
Commercial	15	0.20	0.07	0.59	0.20	33.00	11.22	0.00	0.000
Industrial	16	0.20	0.07	0.59	0.20	33.00	11.22	0.00	0.000
Transitional	17	0.33	0.11	0.89	0.30	80.52	27.38	0.20	0.068
Transportation	18	0.20	0.07	0.59	0.20	33.00	11.22	0.00	0.000
Waste Disposal	19	0.20	0.07	0.59	0.20	33.00	11.22	0.00	0.000
Open Water	20	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.000
Powerline / Utility	24	0.33	0.11	0.89	0.30	80.52	27.38	0.20	0.068
Golf Course	26	0.90	0.31	9.11	3.10	110.00	37.40	0.00	0.000
Marina	29	0.20	0.07	0.59	0.20	33.00	11.22	0.00	0.000
Urban Public / Institutional	31	0.20	0.07	0.59	0.20	33.00	11.22	0.00	0.000
Cemetary	34	0.90	0.31	9.11	3.10	110.00	37.40	0.00	0.000
Orchard	35	5.10	1.73	1.00	0.34	65.00	22.10	1.70	0.578
Nursery	36	0.20	0.07	0.59	0.20	33.00	11.22	0.00	0.000
Forested Wetland	37	49.28	16.76	12.83	4.36	70.29	23.90	20.05	6.817
Very Low Density Residential	38	0.90	0.31	9.11	3.10	110.00	37.40	0.00	0.000
Junkyard	39	0.20	0.07	0.59	0.20	33.00	11.22	0.00	0.000
Shrub/Scrub	40	0.33	0.11	0.89	0.30	80.52	27.38	0.20	0.068

Input data including standard deviation and mean were assigned to the four carbon pools of aboveground biomass, below ground biomass, soil carbon, and dead litter.

The carbon sequestration for each scenario was calculated by multiplying the biophysical characteristics of carbon storage for each pixel with a changed land use class. The InVEST model has the option to either calculate the volume of carbon removed from routine timber harvests or calculate the difference in carbon storage from land use changes. We selected not to utilize the harvested wood products function and forego measuring carbon storage from biomass extraction. Instead we chose to examine land-use changes using the Reducing Emissions from Forest Degradation and Deforestation (REDD) through the scenario comparison option built into the InVEST carbon Model.

The biophysical data for each land use class was adopted from published studies (Tomasso & Leighton, 2014) using the same InVEST model for carbon sequestration calculation in Connecticut. The biophysical parameters were verified through published research (Blumstein & Thompson, 2015; Bridgham, Megonigal, Keller, Bliss, & Trettin, 2006; Buchholz et al., 2014; Canadell et al., 2000; Compton & Boone, 2000; Eggleston, 2006; McFarlane et al., 2013; NCASI, 2016; Post & Kwon, 2000; Raciti et al., 2011; Rao, Hutyra, Raciti, & Finzi, 2013; Tomasso & Leighton, 2014). We performed an uncertainty analysis to accommodate the limitations with accurately estimating the amount of carbon in different pools.

The uncertainty analysis was performed by calculating the standard deviation from normal distribution of carbon pools inputs. We generalized the normal distribution of carbon rates by assuming one standard deviation (34%) from the mean. We selected a confidence threshold of 95% so that the model would only select pixels with a significant p value ($p=0.95$ confidence) indicating that carbon storage will definitively increase or decrease. The uncertainty analysis was computed using a Monte Carlo simulation with

10,000 runs of the model for each scenario. For each run of the Monte Carlo simulation, carbon values in each pixel were drawn independently from the normal distribution of inputs given in the biophysical table. The results from all the Mote Carlo simulations were analyzed to produce the mean and standard deviation of total carbon (Mg/C), carbon sequestered (Mg/C) and Net Present Value ($USD).

The valuation of carbon in the model was based on the social value of sequestered carbon, not the total amount of carbon in the four pools. The social value of carbon is considered to be equivalent to the cost of damage prevented to society from not emitting additional climate-changing GHGs into the atmosphere. The price of carbon in terms of metric tons of CO^2 was set at $13.00 (USD) and was established as the 2015 mean taken from the California Carbon Dashboard (CPI, 2016). The annual rate of price increase for carbon sequestration was selected at 2% with a market discount rate of 3% for the 50 year period of interest. The model outputs were aggregated into a summary table with the total amount of carbon stored and the valuation in dollars for each scenario.

Valuation of Agriculture Products

In order to evaluate the trade-offs from ecosystem serves I estimated the economic value of increased agriculture acreage. I chose to focus on four agriculture revenue streams: dairy (milk) production, beef production, lamb and wool production, and vegetable production. These four production categories were chosen because of the current agriculture infrastructure, climactic conditions and the dietary needs informed by the NEFV. All valuations were based on the wholesale price received by farmers (Keough, 2012; NASS, 2016). As outlined in the NEFV, it was estimated that 60% of the pasture land was devoted to dairy production, 33% devoted to beef production, and the remaining 6% devoted to lamb and wool production.

Dairy production was estimated from the total acreage converted into pasture in each scenario. The land area devoted to dairy includes pasture land for grazing as well as land for hay and haylage production. The dairy herd density was estimated at 1.22 hectares per cow, which includes dry cows, heifers, calves and breeding stock. The active milking cow population was estimated at 70% of the entire dairy herd, assuming a 10 month lactation and an estimated a cull rate of 15% from the dairy herd allocated to beef production. Dairy production estimates for the county were based on a mostly grass-fed ration with supplemental grain. The volume of milk produced was calculated by multiplying the milking heard population by a modest annual production capacity of 6,350 kg/cow/year (Rotz, Roth, & Stout, 2002). The fluid price of milk was estimated at 0.39 $/kg paid to the producer (Blayney & Normile, 2004; NASS, 2016). The market discount rate was calculated at 3% for the 50 year time span of the study.

As mentioned above, the beef cow population was established from the acreage converted to pasture, but limited to the one third of the total pasture area. The beef herd population density was estimated at 1.1 head per hectare (Rinehart, 2006) with 20% cow-calf breed stock. Grass-fed beef production requires a 24 month maturation period, which amounts to 50% of the non-breading stock herd slaughtered annually in addition to15% of the culled dairy herd. The average live weight of the harvested animals was estimated at 499 kg. The USDA estimates on average a 40% yield rate per carcass for grass-finished beef (Holland, 2014). I estimated the average selling price at $13.23/kg wholesale. The market discount rate was set at 3% for the 50 year time span of the study.

I estimated the lamb and wool production capacity on the remaining 6% of the remaining pastureland. The sheep flock density was estimated at 20.6 head/ha with 40% of the flock representing breeding stock ewes and the remaining 60% consisting of lambs for slaughter. I assumed a 10 month maturation process for the meat lambs with an average live weight of 63.5kg and a 40% yield rate. Lamb meat valuation was estimated at $13.23/kg. We calculated wool production from a single annual shearing of the entire flock with an average fleece weight of 3.28kg and a 50% yield rate after cleaning (Schoenian, 2012). I estimated wool valuation at $4.75/kg for cleaned wool. Lamb and wool price estimates were extracted from the USDA Weekly National Lamb Market Summary (A. USDA, 2016).

The fourth agriculture revenue stream we chose to evaluate was vegetable production. As informed by the NEFV dietary guidelines, it is expected that vegetable consumption will increase over the next five decades. I chose to focus exclusively on vegetable production and exclude grain crops from our calculations because grains have not traditionally been grown with great success in Franklin County and are better suited to other regions of the country. The conventional vegetable production was estimated based on the area of cropland converted in each scenario with 30% of the total cropland land in a fallow rotation and or used for livestock forage production. I chose to model the production of four commonly grown crops in New England including snap beans (pole / bush), carrots, head lettuce and fall potatoes. Each of the four crops were allocated one fifth of the total cropland not in fallow rotation. Crop yield per acre and price per pound data was extracted from the 2012 crop summary published by the USDA New England Fruits and Vegetables, 2012 Crop Report (Keough, 2012). The average annual yield per for beans in 2012 across New England was 3,475 kg/ ha; the average wholesale price for beans was $4.19/kg. The average annual yield for carrots was 17,709 kg/ha and sold at an average wholesale price of $2.54/kg. The average annual yield per hectare for head lettuce was 13,338 kg/ha; the average wholesale price for head lettuce was $3.75/kg. The average yield per for fall potatoes was 31,384 kg/ha; the average wholesale price for potatoes was $0.25/kg. The market discount rate was set at 3% for the 50 year time span of the study.

Chapter III

Results

The scenario results confirmed my initial hypothesis that reductions of forested land cover would directly increase the social costs from lost ecosystem services. Reduction in forested land cover reduced nutrient retention and carbon sequestration. Nutrient export increased in each scenario including significant changes over the baseline. Although the study area was focused on land-use changes in Franklin County, the nutrient export results were analyzed in relation to their impact on each complete watershed.

Nitrogen Export

All five watersheds were affected by land-use changes, with the Connecticut and Deerfield experiencing the largest total increases of N pollution. Table 6 shows the total N export for each watershed and the percentage increase over the baseline scenario. Scenario 1 resulted in the largest increase in all five watersheds with a combined N export increase of 82% over the baseline. The Chicopee, Millers and Westfield watersheds experienced the largest increases over the baseline in scenarios 1 and 5 largely due to significant portions of their land area being forested and classified as soil class 6 that were converted into pasture. Increases in total nutrient export under scenarios 5 illustrate the impact of livestock on the landscape because this scenario had no designated cropland. Scenario 4 demonstrated the least amount (5%) of pollution increase over the baseline scenario, which was largely due to the fact that no new cropland was developed and pastureland for livestock production was relatively limited.

Table 6. Total nitrogen export kg/watershed for each scenario.

N export (kg/watershed)											
	Baseline	**Scenario 1**		**Scenario 2**		**Scenario 3**		**Scenario 4**		**Scenario 5**	
Watershed	N export kg/watershed	N export kg/watershed	% increase over baseline	N export kg/watershed	% increase over baseline	N export kg/watershed	% increase over baseline	N export kg/watershed	% increase over baseline	N export kg/watershed	% increase over baseline
Chicopee	14,946	37,207	149%	17,944	20%	16,391	10%	15,396	3%	33,945	127%
Connecticut	114,769	198,006	73%	159,212	39%	133,439	16%	121,336	6%	140,950	23%
Deerfield	81,722	141,996	74%	121,717	49%	104,445	28%	86,865	6%	110,711	35%
Millers	24,986	51,801	107%	37,443	50%	31,917	28%	26,252	5%	41,406	66%
Westfield	29,280	55,480	89%	44,219	51%	34,760	19%	29,382	0%	62,538	114%
Total	**265,703**	**484,490**	**82%**	**380,534**	**43%**	**320,952**	**21%**	**279,231**	**5%**	**389,550**	**47%**

The nutrient exports are relatively high in the baseline scenario for the Connecticut (114,769 kg) and Deerfield (81,722 kg) watersheds, which is indicative of the current impact of agriculture and the concentration of development. These two watersheds account for the largest relative proportion of watershed land area contained within the county boundaries. Across all five scenarios, the Connecticut watershed consistently had the largest concentration of cropland and developed land-use types which was intentionally built into the model design. Selection of new agriculture land was largely precluded from the Connecticut watershed during the scenario creation process because the land area was restricted by current use and thus not suitable for conversion. Although the relative proportion of new agricultural land area in the Connecticut watershed was not as substantial as the other four watersheds, its current agriculture footprint resulted in high levels of nutrient export under the baseline scenario. Additionally, most of the newly converted land in the Connecticut watershed had a significant impact on N export because most of this area was classified as soil class 1, 2 or 3 and consequently converted into cropland with high N export coefficients.

The spatial distribution of newly created cropland in the Connecticut and Deerfield watersheds directly contributed to higher N pollution across scenarios 1, 2 and 3. The concentration of cropland located along the floodplains of the Deerfield and Connecticut Rivers resulted in high export volumes with limited retention capacity for bio-filtration buffering because of the close proximity to waterbodies. The Deerfield and Connecticut watershed also had the largest proportion of land area converted into crop land. The biophysical properties assigned to conventional cropland imply intentional N loading through the application of synthetic fertilizers for increased crop productivity. The increase in N export was a function of the volume of land converted into cropland.

The most noticeable impact on the Westfield watershed can be seen in the maps depicted in figures 19 and 20. Under all scenarios forest clearing was wide scale in this area because of mild elevation gradients characteristic of the eastern Berkshire plateau, which prioritized this land for agriculture expansion based on the scenario decision rules. There is also a heavy agriculture footprint in the western portion of the town of Ashfield contributing to high nutrient export. Most of the land area in Ashfield is classified as soil class 6, which was included in all five scenarios except for scenario 4. Clearing of these marginal lands to create pastureland directly contributed to increased nutrient export in scenarios 1, 2, 3 and 5. It is also notable that the baseline scenario results also showed a high nitrogen export value for the Westfield watershed, which may be caused by a modeling error pictured in Figure 19. Numerous runs of the model with different input data yielded high results in this portion of the county, which is why we have chosen to include these results in Figure 19 and 20.

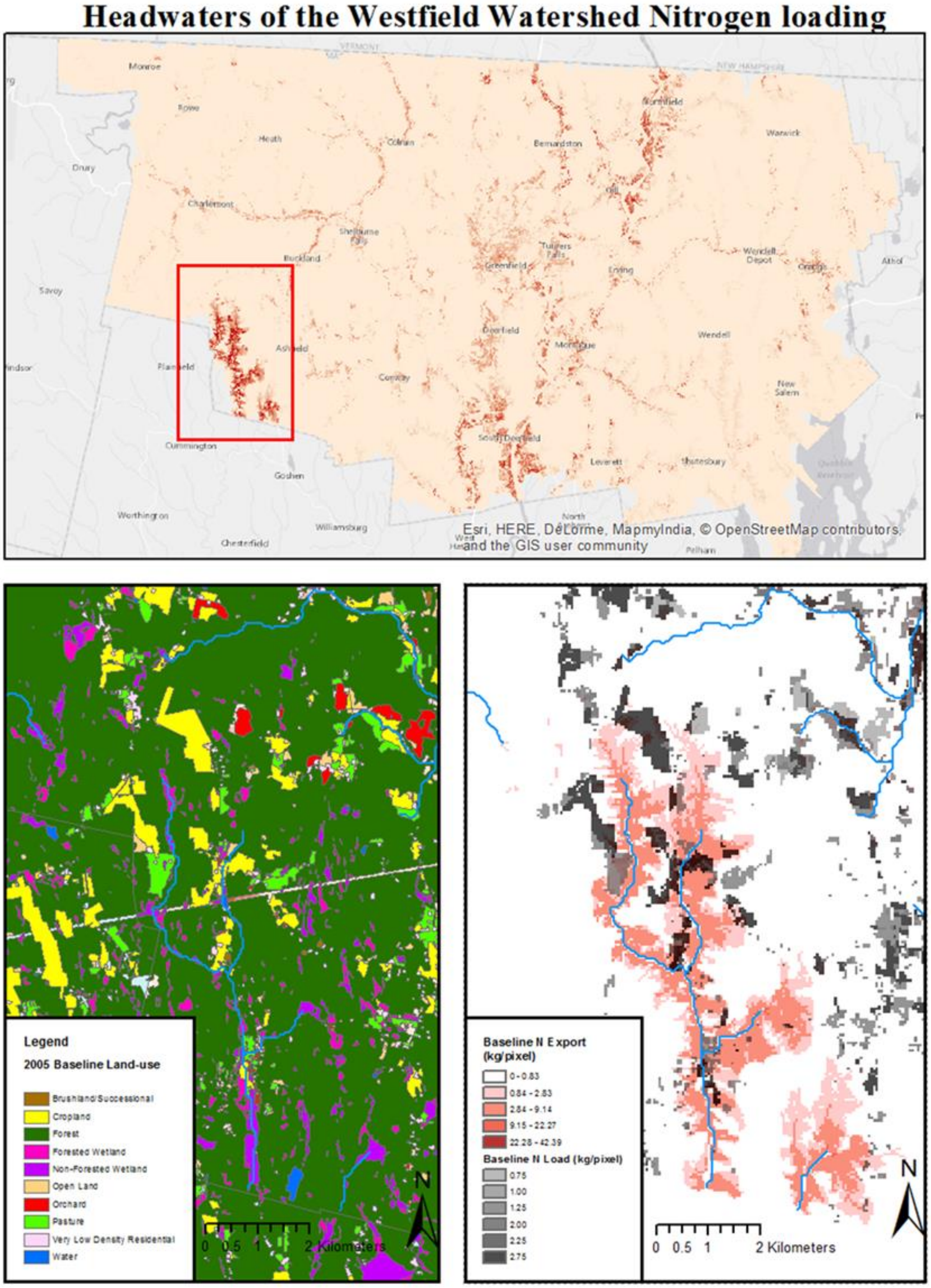


Figure 19. Nitrogen loading and export for the across the headwaters of the Westfield watershed under the baseline scenario. Note the loading of nitrogen from cropland depicted in yellow on the lower left and as dark grey on the lower right. Total export per pixel is shown in red on top and lower right maps.

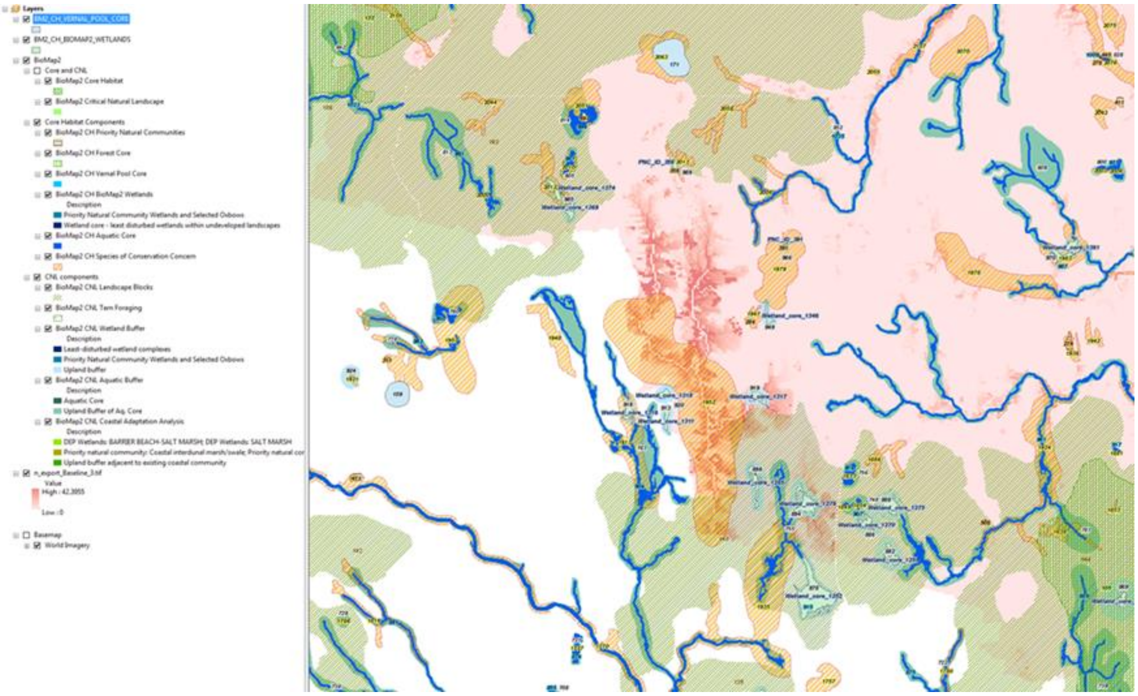


Figure 20. Nitrogen export in the headwaters of Westfield watershed overlaid by BioMap2 conservation features. Note the lack of stream buffers for this area and incomplete coverage of the aquatic core for the Westfield stream network. Red color scale shows nitrogen export into the Westfield watershed.

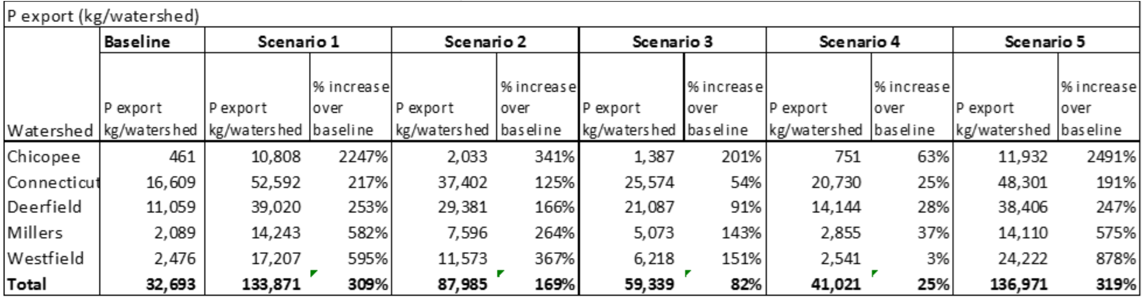
Phosphorus Export

The increase in P pollution was significantly higher than the baseline for all five watersheds. There was a significant increase in total P export across all five watersheds for each of the scenarios. Total P export increased from 32.7 MT in the baseline to 137 MT in scenario 5. The biophysical properties of pastureland specify very high P export coefficients, which is why new land area converted into pasture translates into higher P pollution. The large increase in P export under scenarios 4 and 5 are almost exclusive to new pasture creation because both of these scenarios had no new cropland created.

The largest impacts from forest conversion to pasture resulted in very high nutrient exports along the Westfield and Chicopee rivers. For scenario 1, P export in the

Chicopee watershed increased by a factor of 23.47 and by a factor of 25.9 for scenario 5. The Westfield watershed also demonstrated significant increases in P export above the baseline scenario for scenario 1 (595%) and scenario 5 (878%). This large increase in these two watersheds was the result of limited conservation restrictions in these areas and the selection of marginal soil class for conversion to pasture from forests in the scenario creation process.

Table 7. Total phosphorus export kg/watershed for each scenario.



P export (kg/watershed)

	Baseline	Scenario 1		Scenario 2		Scenario 3		Scenario 4		Scenario 5	
Watershed	P export kg/watershed	P export kg/watershed	% increase over baseline	P export kg/watershed	% increase over baseline	P export kg/watershed	% increase over baseline	P export kg/watershed	% increase over baseline	P export kg/watershed	% increase over baseline
Chicopee	461	10,808	2247%	2,033	341%	1,387	201%	751	63%	11,932	2491%
Connecticut	16,609	52,592	217%	37,402	125%	25,574	54%	20,730	25%	48,301	191%
Deerfield	11,059	39,020	253%	29,381	166%	21,087	91%	14,144	28%	38,406	247%
Millers	2,089	14,243	582%	7,596	264%	5,073	143%	2,855	37%	14,110	575%
Westfield	2,476	17,207	595%	11,573	367%	6,218	151%	2,541	3%	24,222	878%
Total	**32,693**	**133,871**	**309%**	**87,985**	**169%**	**59,339**	**82%**	**41,021**	**25%**	**136,971**	**319%**

Nutrient Export Valuation

Valuation of nutrient export was analyzed with two alternative treatment methods, the Nutrient Management Plans (NMP) and Waste Water Treatment (WWT) facilities. These pricing strategies were intended to illustrate the difference in cost to society to remediate the impact from excessive nutrient export. The WWT costs assigned to forgone nutrient retention are considerably higher than those of the NMP. Export costs were highest in the Connecticut and Deerfield watersheds for all of the scenarios as these scenarios had the largest overall export volume. The Connecticut watershed consistently had the largest nutrient export value except for P valuation under scenario 3, which resulted in the Deerfield having slightly higher values. This is largely due to the conservation of limited areas of prime cropland under BioMap2 for conservation of rare species and habitats. The combined N and P nutrient retention costs ranged from $33.3 to $139.9 million under the NMP costing assumptions. The WWT costing assumptions produced considerably higher outputs ranging from $118.9 to $423.2 million. Comparison in the costs between the two remediation strategies can be seen in Tables 8 and 9. Maintaining the baseline scenario into the future accounted for the least amount of nutrient loading. Total remediation costs for each scenario can be seen Table 10.

Table 8. Valuation of nitrogen export with nutrient management plan vs waste water treatment costs in 2060.

Cost of N remediation: Nutrient Management Plan vs. Waste Water Treatment costs NPV 2060 (USD)												
	Baseline		**Scenario 1**		**Scenario 2**		**Scenario 3**		**Scenario 4**		**Scenario 5**	
Watershed	N Remediation NMP	N Remediation WWT	N Remediation NMP	N Remediation WWT	N Remediation NMP	N Remediation WWT	N Remediation NMP	N Remediation WWT	N Remediation NMP	N Remediation WWT	N Remediation NMP	N Remediation WWT
Chicopee	$ 1,278,798	$ 6,615,649	$ 2,673,270	$ 13,829,715	$ 1,538,388	$ 7,958,593	$ 1,387,448	$ 7,177,729	$ 1,304,528	$ 6,748,756	$ 2,614,234	$ 13,524,306
Connecticut	$ 4,510,407	$ 23,333,840	$ 7,668,620	$ 39,672,325	$ 6,136,970	$ 31,748,590	$ 4,946,795	$ 25,591,419	$ 4,609,636	$ 23,847,183	$ 5,405,083	$ 27,962,295
Deerfield	$ 3,377,463	$ 17,472,742	$ 5,798,253	$ 29,996,298	$ 4,931,051	$ 25,509,972	$ 4,279,832	$ 22,140,997	$ 3,501,319	$ 18,113,490	$ 4,688,789	$ 24,256,670
Millers	$ 1,181,838	$ 6,114,040	$ 2,339,479	$ 12,102,905	$ 1,693,708	$ 8,762,119	$ 1,475,912	$ 7,635,387	$ 1,228,658	$ 6,356,255	$ 1,818,368	$ 9,407,025
Westfield	$ 2,109,041	$ 10,910,773	$ 3,780,543	$ 19,558,007	$ 3,064,901	$ 15,855,757	$ 2,455,378	$ 12,702,487	$ 2,126,311	$ 11,000,114	$ 3,523,592	$ 18,228,718
Total	**$ 12,457,547**	**$ 64,447,044**	**$ 22,260,164**	**$ 115,159,249**	**$ 17,365,019**	**$ 89,835,030**	**$ 14,545,364**	**$ 75,248,019**	**$ 12,770,451**	**$ 66,065,798**	**$ 18,050,067**	**$ 93,379,014**

Table 9. Valuation of phosphorus export with nutrient management plan vs waste water treatment costs in 2060.

Cost of P remediation: Nutrient Management Plan vs. Waste Water Treatment costs NPV 2060 (USD)												
	Baseline		**Scenario 1**		**Scenario 2**		**Scenario 3**		**Scenario 4**		**Scenario 5**	
Watershed	P Remediation NMP	P Remediation WWT	P Remediation NMP	P Remediation WWT	P Remediation NMP	P Remediation WWT	P Remediation NMP	P Remediation WWT	P Remediation NMP	P Remediation WWT	P Remediation NMP	P Remediation WWT
Chicopee	$ 627,888	$ 1,644,165	$ 14,042,836	$ 36,772,076	$ 3,615,847	$ 9,468,330	$ 2,224,314	$ 5,824,510	$ 961,508	$ 2,517,772	$ 15,402,216	$ 40,331,698
Connecticut	$ 9,183,493	$ 24,047,571	$ 39,089,753	$ 102,359,049	$ 26,053,770	$ 68,223,482	$ 14,427,290	$ 37,778,792	$ 10,931,010	$ 28,623,556	$ 30,578,082	$ 80,070,687
Deerfield	$ 5,917,993	$ 15,496,647	$ 30,604,577	$ 80,140,067	$ 21,634,202	$ 56,650,558	$ 14,511,887	$ 38,000,316	$ 7,712,880	$ 20,196,676	$ 27,449,574	$ 71,878,486
Millers	$ 1,509,255	$ 3,952,082	$ 12,448,323	$ 32,596,740	$ 6,180,729	$ 16,184,639	$ 4,108,374	$ 10,758,044	$ 2,123,984	$ 5,561,789	$ 10,892,850	$ 28,523,634
Westfield	$ 3,581,285	$ 9,377,827	$ 21,461,731	$ 56,198,932	$ 14,850,089	$ 38,885,919	$ 8,318,817	$ 21,783,360	$ 3,816,243	$ 9,993,080	$ 23,751,424	$ 62,194,641
Total	**$ 20,819,914**	**$ 54,518,292**	**$ 117,647,221**	**$ 308,066,865**	**$ 72,334,636**	**$ 189,412,929**	**$ 43,590,682**	**$ 114,145,022**	**$ 25,545,625**	**$ 66,892,873**	**$ 108,074,145**	**$ 282,999,147**

Table 10. Total remediation costs of nitrogen and phosphorus for each scenario comparing nutrient management plan vs waste water treatment costs in 2060.

Total remediation cost N and P: Nutrient Management Plan vs. Waste Water Treatment NPV 2060 (USD)						
	Value N (NMP)	Value N (WWT)	Value P (NMP)	Value P (WWT)	**TOTAL N and P (NMP)**	**TOTAL N and P (WWT)**
Baseline	$ 12,457,547	$ 64,447,044	$ 20,819,914	$ 54,518,292	**$ 33,277,461**	**$ 118,965,336**
Scenario 1	$ 22,260,164	$ 115,159,249	$ 117,647,221	$ 308,066,865	**$ 139,907,385**	**$ 423,226,115**
Scenario 2	$ 17,365,019	$ 89,835,030	$ 72,334,636	$ 189,412,929	**$ 89,699,655**	**$ 279,247,959**
Scenario 3	$ 14,545,364	$ 75,248,019	$ 43,590,682	$ 114,145,022	**$ 58,136,046**	**$ 189,393,041**
Scenario 4	$ 12,770,451	$ 66,065,798	$ 25,545,625	$ 66,892,873	**$ 38,316,076**	**$ 132,958,672**
Scenario 5	$ 18,050,067	$ 93,379,014	$ 108,074,145	$ 282,999,147	**$ 126,124,213**	**$ 376,378,161**

Carbon Sequestration and Valuation

Carbon sequestration results indicate the forgone value of sequestered carbon. Table 11 shows the total amount of carbon stored and sequestered as well as the value of sequestered carbon. The baseline scenario is the only scenario to show a net increase in the amount of carbon sequestered because it was assumed in the scenario creation process that the baseline would retain the largest land area as forested.

Table 11. Total carbon storage and sequestration for each scenario with NPV in 2060.

Total Carbon stored, sequestered and value of sequestered carbon NPV 2060			
	Total Carbon Stored Mg/C	Total Carbon Sequestered Mg/C	Value of Sequestered C (USD)
Baseline	35,272,699	1,114,235	$ 18,709,841
Scenario 1	19,832,724	(15,488,575)	$ (260,078,794)
Scenario 2	26,829,169	(8,423,138)	$ (141,438,427)
Scenario 3	31,279,383	(3,989,785)	$ (66,995,089)
Scenario 4	34,162,237	(1,108,648)	$ (18,616,037)
Scenario 5	21,360,031	(13,881,389)	$ (233,091,494)

All of the scenarios with future agriculture expansion had a net loss in carbon due to deforestation. The uncertainty analysis results reflect the range of variability in the response of sequestration rates. Figure 21 shows the amount of sequestered carbon for each scenario including the Monte Carlo uncertainty analysis with one standard deviation from the mean. The valuation of sequestered carbon in scenarios 1 through 5 illustrates the lost revenue to society assuming that the carbon credits could be sold in a cap and trade type valuation system.

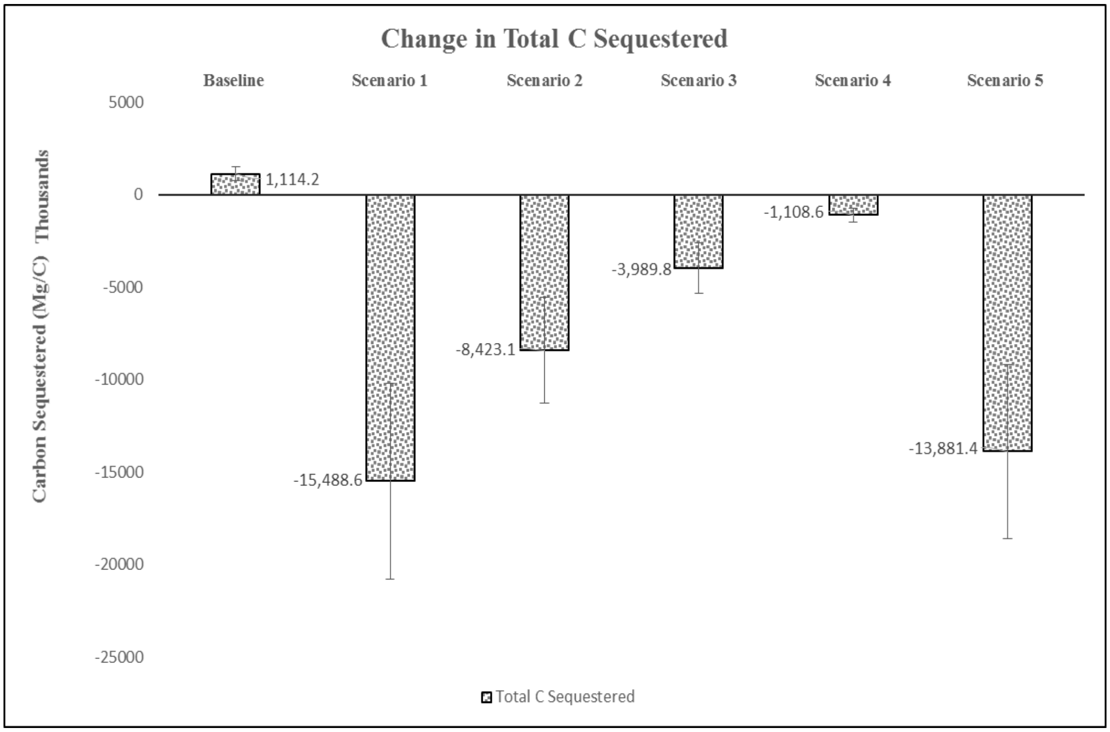


Figure 21. Change in total carbon sequestered (Mg/C) for each scenario in 2060.

Agriculture Production and Valuation

The value of agriculture products for each scenario was comprised of four livestock production systems for pastureland comprising dairy, beef, lamb and wool; and vegetable production on cropland consisting of the four crop types of beans, carrots, head lettuce and potatoes. Livestock and vegetable capacity was based on the allocation of cropland and pasture created in each scenario. The model assumes that agriculture activity will be maximized on the baseline land-use matrix before new land is developed for agriculture. Increased vegetable production from the four crops of beans, carrots, head lettuce and potatoes yielded the greatest value. This increase in vegetable valuation is concurrent with NEFV and the projected trends toward greater consumption of fresh vegetables. Each scenario assumes farmers will continue to convert marginal lands (soil class 3) into vegetable production. Table 12 illustrates the estimated value of vegetable production in the year 2060.

Table 12. Total vegetable production valuation for each scenario.

		Hectares in production	**Beans** (Snap, Pole, Bush)		**Carrots**		**Head Lettuce**		**Fall Potatoes**		**Total Veg Value**	
	Hectares Cropland	30% fallow rotation used for hay and livestock feed	NE Average yeild per hectare = 3475 kg	Average price per kg = $4.19	NE Average yeild per hectare = 17709 kg	Average price = $2.54/kg	NE Average yeild per hectare = 13338 kg	Average price = $3.75/kg	NE Average yeild per hectare = 31384	Average price per kg = $0.25	Discount Rate = 3% *50 years	% change over baseline
Baseline	9,613	6,729	5,845,283	$ 24,491,735	29,792,087	$ 75,531,879	22,438,344	$ 84,143,791	52,796,104	$ 13,199,026	$ 45,020,680	
Scenario 1	29,891	20,924	18,175,739	$ 76,156,345	92,637,636	$ 234,864,199	69,771,384	$ 261,642,691	164,167,963	$ 41,041,991	$ 139,990,507	211%
Scenario 2	22,291	15,604	13,554,340	$ 56,792,685	69,083,411	$ 175,147,171	52,031,176	$ 195,116,912	122,426,297	$ 30,606,574	$ 104,396,248	132%
Scenario 3	16,791	11,754	10,209,795	$ 42,779,043	52,037,022	$ 131,929,462	39,192,441	$ 146,971,653	92,217,507	$ 23,054,377	$ 78,636,388	75%
Scenario 4	9,473	6,631	5,760,338	$ 24,135,818	29,359,144	$ 74,434,237	22,112,266	$ 82,920,999	52,028,862	$ 13,007,216	$ 44,366,432	-1%
Scenario 5	N/A											

Value estimated for beans, carrots, head lettuce and potatoes in the year 2060.

The combined value of agriculture products increased by 275% over the baseline in scenario 1 to a modest increase of 6% in scenario 4. Dairy, beef and lamb valuation does not experience the same rate of growth as vegetable production. Scenario 3 resulted in an 89% increase over the baseline, while scenario 4 only increased production values by 6%. Table 13 demonstrates the combined value of agriculture output.

Table 13. Combined valuation of agriculture products.

Agriculture product valuation (USD) NPV 2060						
	Dary valuation	Beef valuation	Lamb and wool valuation	Vegetable Valuation	Total value	% increase over baseline
Baseline	$ 996,730	$ 615,561	$ 353,911	$ 45,020,680	**$ 46,986,882**	
Scenario 1	$ 18,311,837	$ 11,309,047	$ 6,502,018	$ 139,990,507	**$ 176,113,408**	275%
Scenario 2	$ 10,281,680	$ 6,349,773	$ 3,650,735	$ 104,396,248	**$ 124,678,437**	165%
Scenario 3	$ 5,089,779	$ 3,143,352	$ 1,807,238	$ 78,636,388	**$ 88,676,757**	89%
Scenario 4	$ 2,678,186	$ 1,653,997	$ 950,948	$ 44,366,432	**$ 49,649,564**	6%
Scenario 5	$ 26,197,223	$ 16,178,914	$ 9,301,897	$ -	**$ 51,678,034**	10%

Values include dairy, beef, lamb, wool, and vegetables for each scenario in the year 2060.

Combined Valuation with Waste Water Treatment Costs

The net impact from deforestation on ecosystem services outweighs the increased value from agriculture products. The ecosystem services provided by forests to reduce N and P runoff nearly outweighs all of the gains from agriculture production when considering the higher remediation costs from WWT. Scenario 5 resulted in the greatest loss of value to society (-$557 million), which is part due to the lack of a highly valuable vegetable production system. Scenario 1 had the most robust agriculture output worth $176 million, but the second largest loss of net value to society. Table 14 and Figure 22 show the net value for each scenario of agriculture products and ecosystem services with the WWT costs.

Table 14. Net value to society from combined ecosystem services and agriculture products with waste water treatment costs.

Combined Valuation with WWT (USD) NPV 2060						
	Baseline No-Change	**Scenario 1**	**Scenario 2**	**Scenario 3**	**Scenario 4**	**Scenario 5 - 1830's**
Ecosystem Service Valuation						
Combined N and P **(WWT)** Cost to Remediate	$118,965,336	-$423,226,115	-$279,247,959	-$189,393,041	-$132,958,672	-$376,378,161
Carbon Sequestration - Forgone C sequestration value	$18,709,841	-$260,078,794	-$141,438,427	-$66,995,089	-$18,616,037	-$233,091,494
Agriculture valuation						
Pastured Dairy - Milk	$996,730	$18,311,837	$10,281,680	$5,089,779	$2,678,186	$26,197,223
Pasture Beef - Grass-fed beef	$615,561	$11,309,047	$6,349,773	$3,143,352	$1,653,997	$16,178,914
Pastured Sheep - Lamb and wool	$353,911	$6,502,018	$3,650,735	$1,807,238	$950,948	$9,301,897
Cropland Cultivated - Vegetables	$45,020,680	$139,990,507	$104,396,248	$78,636,388	$44,366,432	
Net Value	$184,662,058	-$507,191,500	-$296,007,949	-$167,711,372	-$101,925,145	-$557,791,621

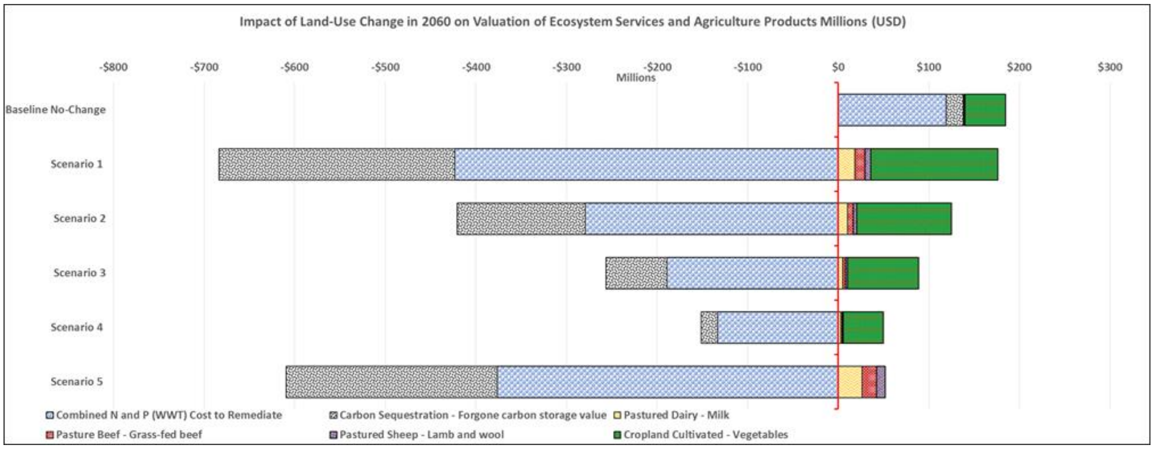
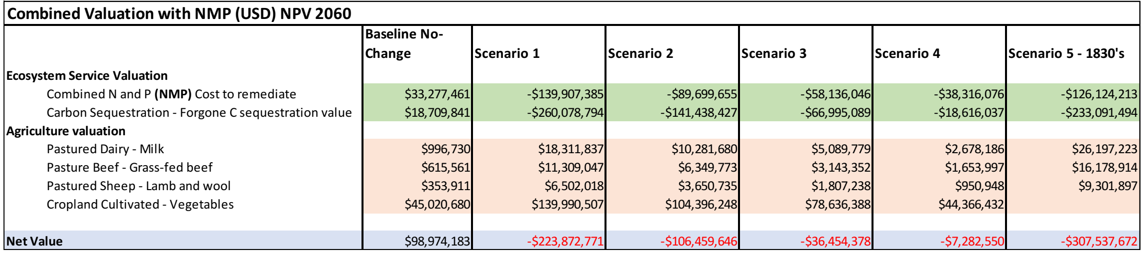


Figure 22. Net present value of ecosystem services with waste water treatment costs and agriculture products.

Combined Valuation with Nutrient Management Plan Costs

Under scenarios 1, 2, 3 and 5 the combined social cost of decreased carbon sequestration and increased nutrient loading into the lower Connecticut drainage basin outweighs the benefits of increased production. The baseline land-use pattern projects the largest net increase in both agriculture productivity and ecosystem services. Although less severe than the WWT, the alternative NMP approach resulted in significant net losses to society ranging from -$7.2 million for scenario 4 to -$307.5 million under scenario 5. Table 15 and Figure 23 show the balance of ecosystem service costs and product values. The NMP method is clearly more cost effective than WWT, but the social value of forested land when combined with the value of carbon sequestration easily outweighs the sum of agriculture products.

Table 15. Net value to society from combined ecosystem services and agriculture products with nutrient management plan costs.



Combined Valuation with NMP (USD) NPV 2060						
	Baseline No-Change	**Scenario 1**	**Scenario 2**	**Scenario 3**	**Scenario 4**	**Scenario 5 - 1830's**
Ecosystem Service Valuation						
Combined N and P **(NMP)** Cost to remediate	$33,277,461	-$139,907,385	-$89,699,655	-$58,136,046	-$38,316,076	-$126,124,213
Carbon Sequestration - Forgone C sequestration value	$18,709,841	-$260,078,794	-$141,438,427	-$66,995,089	-$18,616,037	-$233,091,494
Agriculture valuation						
Pastured Dairy - Milk	$996,730	$18,311,837	$10,281,680	$5,089,779	$2,678,186	$26,197,223
Pasture Beef - Grass-fed beef	$615,561	$11,309,047	$6,349,773	$3,143,352	$1,653,997	$16,178,914
Pastured Sheep - Lamb and wool	$353,911	$6,502,018	$3,650,735	$1,807,238	$950,948	$9,301,897
Cropland Cultivated - Vegetables	$45,020,680	$139,990,507	$104,396,248	$78,636,388	$44,366,432	
Net Value	$98,974,183	-$223,872,771	-$106,459,646	-$36,454,378	-$7,282,550	-$307,537,672

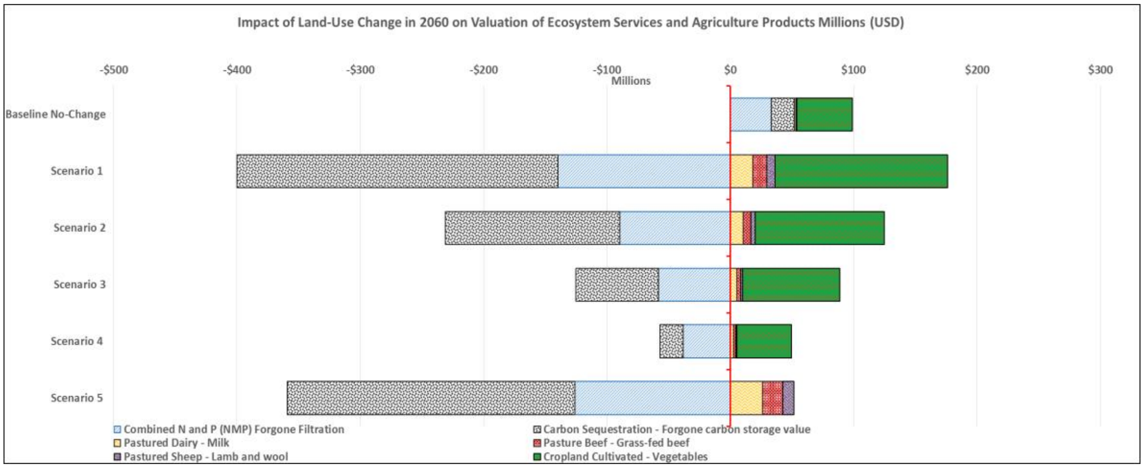


Figure 23. Net present value of ecosystem services with nutrient management plan values and agriculture products.

Chapter IV

Discussion and Conclusion

Model outputs highlight the significance of forested land cover for providing ecosystems services. Conservation of forests has a greater social value in each scenario and far outweighs the trade-off value from establishing new agriculture lands in the county.

Interpretation of Nutrient Retention Results

The results suggest that mitigation of eutrophication from agriculture expansion into forested areas should be discouraged. Additionally, future conservation plans outlined in BioMap2 should be pursued because they are unlikely to reduce the overall value of agriculture production while providing the greatest benefit to society by reducing the impacts from increased nutrient loading. Further, the results suggest that BioMap2 should be revised to establish aquatic and wetland buffers in the headwaters of the Westfield River watershed. The nitrogen export model results for all scenarios, including the baseline, suggest that crop production in the western portion of the town of Ashfield is contributing to heavy N loading. We believe this is due to the relatively large consolidated size of crop production in the area and unique topography. Although this is an area of concern, more research is needed to confirm these findings. Some of the "noise" generated in this portion of Westfield watershed in Franklin County is probably due to deficiencies in model design because multiple runs projected similar results. We advise that future research use the USGS SPARROW stream monitoring data and models to calibrate and modify the InVEST nitrogen retention model. Nevertheless, these results strongly suggest the need for additional research and monitoring as well a comprehensive review of future conservation goals in the headwaters of the Westfield River.

Mapping nutrient loads and transport at the major watershed boundary would have been logically appropriate and technically more convenient than mapping at the county political boundary. Several sub-watersheds were contained within the county or in direct contact with the county boundary, but this does not accurately reflect the entire drainage basin impact, and prematurely limits the scope of inference. During the research design process it was deemed appropriate to focus on the county because GIS calculations and land-use changes were made at the county level, but after analyzing the results I concluded that the study area should be focused on the watershed boundary even though the county boundary is the standard level for USDA agriculture production and soil data. Other challenges arise when focusing on the watershed level as opposed to the county level because the discontinuity of NRCS soil data available for the state. Franklin County soil data was published at a later date (2012) and is the most current, precise and accurate of all the counties of Massachusetts. Limiting the study area to the county boundary is not conducive to modeling hydrological processes, but it is the most conducive for analyzing land use changes based on soil attributes. Since all five watersheds drain into the Connecticut River drainage basin, the overall impact of nutrient pollution should be measured for the entire drainage basin. The future analysis of nutrient retention focus on the Lower Connecticut Major Drainage Basin should allow for a more complete analysis.

I also have reservations as to the accuracies of the N and P export coefficients first published by Reckhow in 1980 (Reckhow et al., 1980). Agriculture methods have changed considerably over the past four decades including the adoption of soil conservation practices such as perennial buffer strips, seasonal manure management, cover cropping and organic farming practices. Additionally, precision agriculture techniques now incorporate routine soil testing for need-specific application of chemical soil amendments. These techniques have become widely employed across the Franklin County over the past four decades, which question the N and P export coefficients used in my model. Considerably more research is needed to determine the accuracy of nutrient export coefficients for contemporary practices and to examine the feasibility in their application on a landscape scale.

The valuation of N and P export is most likely inaccurate for Franklin County, because valuation numbers were adopted directly from estimates derived for the Chesapeake Bay, which has a different topography and land use pattern. There are no Total Maximum Daily Load (TMDL) estimates published for the five watershed analyzed in this study, which would be useful for calculating the cost of eutrophication downstream. Although the N and P valuation figures from the Chesapeake Bay are most likely inaccurate for direct application to Franklin County, they provide valuable insight to the consequences of landscape scale agriculture expansion. For the purposes of this study I felt these valuations were appropriate, but greater consensus is needed amongst the scientific community for valuing the ecosystem service of nutrient retention. Policy measures designed to avoid deforestation will provide the greatest security for ecosystem services, but they will need to be balanced against the demand for food production.

Reduced nutrient loading sources upstream have the potential to drastically improve the cost effectiveness of remediation as well as the deterred social costs of polluted waterways.

Interpretation of Carbon Model Results

The carbon storage and sequestration results reflect the direct correlation between forest cover and the percent converted to crop or pastureland. The InVEST carbon model results are useful when comparing ecosystem services in the cost benefit analysis between trade-off scenarios, but the model is not sophisticated enough for establishing cap and trade mechanisms or complete carbon inventories. We also did not quantify the volume of harvested wood even though the InVEST carbon storage module has this option. Future research into the carbon storage capacity of the region will most likely involve more complex modeling and the wealth of data available from the Harvard Forest research facility.

The InVEST carbon model oversimplifies the rate of sequestration by assuming a linear rate of carbon accumulation over time, which discounts the actual rate of sequestration experienced in successional and maturing forests. Long term carbon storage research at the Harvard Forest has demonstrated that carbon sequestration does not occur at a fixed linear rate, but rather an asymptotic curve. This research also suggests that similar forest types found in Franklin County are continuing to sequester carbon at a rate exceeding linear growth and that the stage carbon storage equilibrium is decades if not centuries beyond 2060 (Thompson et al., 2011). The InVEST carbon model proved to be too simplistic for complete carbon modeling when compared to published research examining the same region. For these reasons we recommend that more sophisticated models be used in future research to more accurately analyze the tradeoffs.

The valuation of carbon has the potential to increase significantly if the policy atmosphere progresses in the coming years. We intentionally chose a low discount rate of 3% to provide a conservative estimate of economic value; however, there is an ongoing debate within the scientific community regarding the most appropriate discount rate for ecosystem services. The market discount rate is often too high in many ecosystem service analyses because the value of money today surpasses that of nearly any future ecosystem service value when compared to the short term benefits from development. By selecting a low discount rates in this project, we were able to equally assess the benefits of both ecosystem services and agriculture products.

The pricing of sequestered carbon also has the potential to increase significantly from the 2015 value of $13.00 if the United States adopts more progressive GHG mitigation policies. This is a likely outcome given the Paris Climate agreement in December of 2015. The forests of New England are in a unique position to take advantage of potential cap and trade systems, but they need to be conserved from development and unwarranted agriculture expansion. Carbon credits could provide the necessary incentive to reduce suburban sprawl. Policy needs to promote forest conservation as well as incentivize the research and deployment of carbon sequestering agriculture practices. Our analysis does not account the GHG emissions from increased livestock production or other agriculture sources like methane from manure, nitrous oxide from synthetic fertilizer and carbon dioxide from farm equipment and transportation. While local production efficiencies may be less than that of imported food, the transportation costs of foregone carbon storage need to be analyzed against the benefits derived from promoting reforestation over agriculture production in future research.

Modeling the negative and positive feedback loops from climate change was beyond the scope of this project, but it is widely accepted that the growing season will be extended in Franklin County from a relatively warmer climate (hardiness zone). Changes to the climate will also impact forest growth rates, microbial decomposition activity and total carbon storage capacity. The frequency and intensity of extreme weather events including severe storms, droughts and flooding will most likely diminish some of the region's crop production capacity. Any effort to model future climactic changes on a landscape scale will be of high importance policy makers, as well as immediately useful for regional farmers and agronomist as tangible knowledge valuable for developing hybrid crop cultivars.

Valuation of Agriculture Products

The valuation of agriculture products is not meant to be comprehensive, but to elucidate the net benefits from agriculture expansion based on the dietary guidelines of the NEFV. I did not estimate the value of value-added food products that can substantially improve the revenue to farmers. I also did not estimate the value difference between direct sales and wholesale markets, but used the USDA market summary data which is biased to wholesale markets. These factors most likely undervalue agriculture products under the different scenarios, but for the scope of this study they were deemed appropriate and sufficient.

There are several production systems that are critical to the agriculture economy of Franklin County that were not considered including wood and timber forest products, maple syrup, fruit orchards, animal feed (hay), swine, and poultry. These were not incorporated into the model deliberately because they do not have a clear spatial determinant like that of vegetables, sheep and bovine. Swine and poultry rely heavily on imported grains so they were not considered into the landscape production potential, but they play an important role in the region's economy, soil fertility and nutrient export. Importing food and feed resources is a necessity in New England that too often defers the environmental impacts to other regions and landscapes. Future analysis should evaluate the impact on ecosystem services of imported products.

Franklin County is an ideal location for apple orchard expansion, but spatially predicting the location of orchards based on soil type and land use patterns was beyond the scope of this project, however any future analysis will need to take into consideration the value of fruit orchards. The model design was limited to single land-use designations, which is an oversimplification of modern agriculture in the region. This made it particularly difficult to identify future orchards based on the model inputs because the distribution of soil classes in the county was uneven across forested land available for expansion. It is also unclear as to how best assign carbon sequestration rates for orchards and what the nutrient export coefficients exist. Future analysis should develop more complex methodologies that incorporate fruit orchard production benefits and impacts.

Conclusion

Our results suggest that achieving the NEFV will require significant sacrifices to ecosystem services in rural communities like Franklin County. Although Franklin County could realistically meet the nutritional demands of all the county residents under scenario 4 by modestly expanding pastureland by 7,404 ha (18,296 a) as well as attaining future conservation goals. Previous studies have demonstrated the County's capacity to feed all of its current residents under a NEFV "omnivore's delight" scenario (Lane, 2012; Sloan et al., 2015), but to develop a regional foodshed as described by the NEFV considerably more farm land would be needed. Further analysis is needed to determine if these findings can be widely applied across New England; however, this study suggests that rural communities barely have the capacity to feed themselves, let alone the major population centers of the region.

In order to produce 50% of the region's food demands by 2060, rural communities like Franklin County will need to be major net exporters to regional population centers. But my results reveal the limited availability of unrealized prime farmland (soil classes 1, 2, and 3) and the extent to which agriculture expansion would need to occur on marginal soil types. Further, these results indicate the negative social costs from compromised ecosystem services and the relative degree to which agriculture expansion impacts these services.

Mixed land use types developed for long term agro-forestry management were not analyzed in this project, but we can speculate that they could provide the greatest value to society for food and timber production while maintaining ecosystem services. The

InVEST model design proved to be oversimplified and is not currently capable of modeling silvo-pasture techniques and multi-crop mixed land-uses. Beyond the inadequacies of current GIS models, there is little to no published data to establish coefficients that accurately represent complex agro-forestry techniques. Field research is needed to determine the carbon storage rates and nutrient retention capacity of these systems. It is also unclear how much food, fuel and fiber these systems can typically produce because standardization of these techniques is yet to be proven on a landscape scale. Even though there is increasing interest in agro-ecology techniques by farmers and academics, the authors of the NEFV have also chosen to exclude their potential impacts and benefits from their regional long term food vision. Despite these shortcomings, agro-forestry techniques have the potential to be the optimal land use policy leading towards sustainability.

Designing sustainable and effective policy instruments that achieve multiple goals simultaneously will be a major challenge in the 21st century. Developing local food systems has the potential to revitalize rural economies while minimizing public health risks through improved nutrition, but these aims need to be balanced against the need to preserve necessary ecosystems services. Policy makers will need to focus economic and financial instruments toward ecologically appropriate agriculture production types that improve public health and ecosystem services by incentivizing farmers. Likewise, incentives that modify consumer behavior towards choosing locally produced food can create numerous economic benefits in the form of reduced nutrition related diseases, reduced health care costs, improved rural economies, and the preservation of rural landscapes. The impetus for county level policy change will need to occur at the town and state level and will most likely be driven by local land-trusts and non-profit organizations because the Franklin County governance structure was dissolved by the state in 1997. The Franklin Regional Council of Governments (FRCOG) has emerged as an alternative means for towns to satisfy common policy needs and goals. The FRCOG is actively engaging county farmers and residents to pursue the NEFV as well as land conservation goals. These results and future research efforts should encourage analysis of alternative agriculture techniques on forested and current farmland in the county. Ultimately, conservation efforts will need to be balanced with the need for food production that is both economically viable and ecologically sustainable.

Appendix 1

Nitrogen and Phosphorus Export under Different Scenarios

Figure 24. Relative changes in nitrogen export for each watershed. Note that each watershed drains into the Connecticut River Drainage Basin.

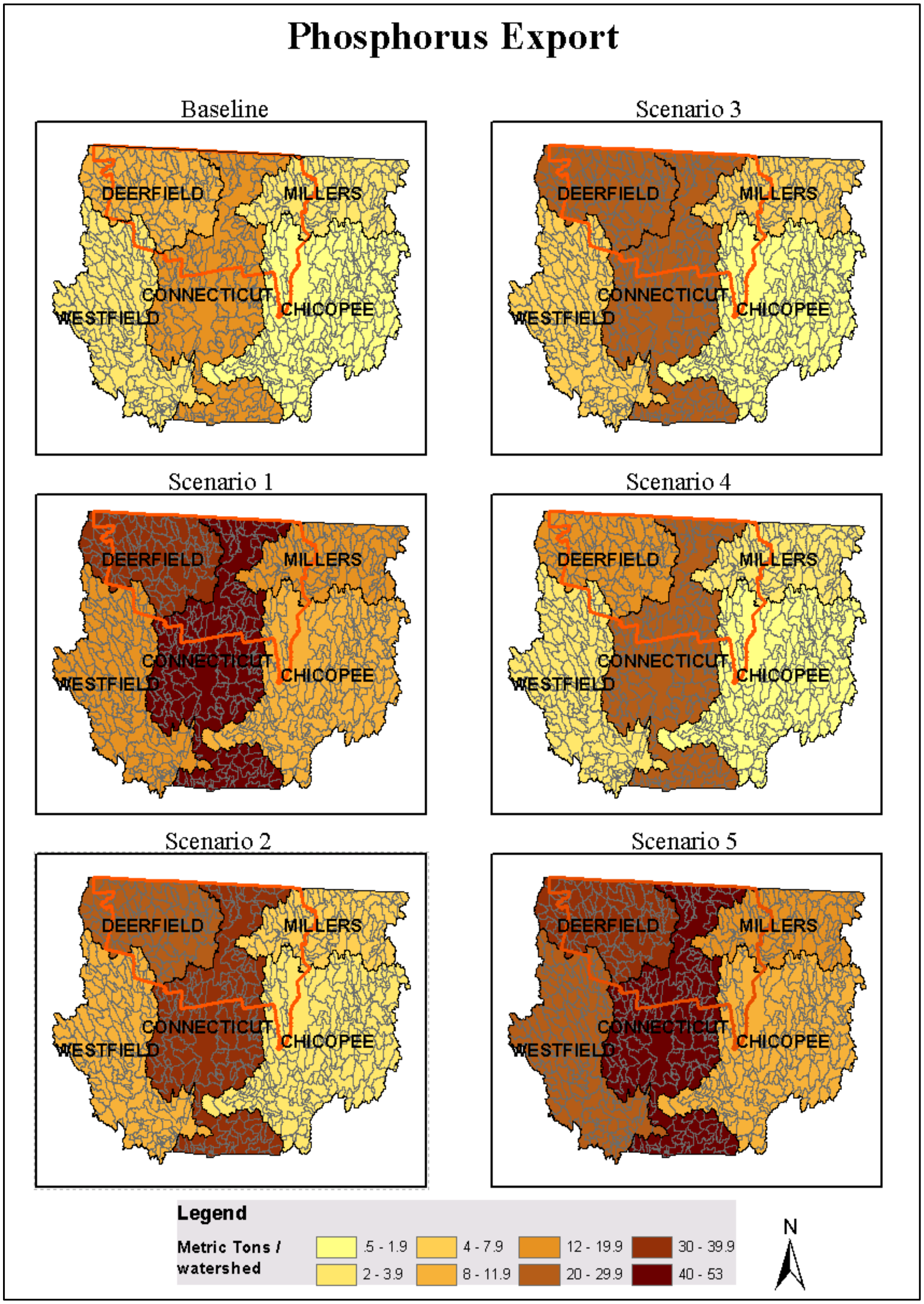


Figure 25. Relative changes in phosphorus export for each watershed. Note that each watershed drains into the Connecticut River Drainage Basin.

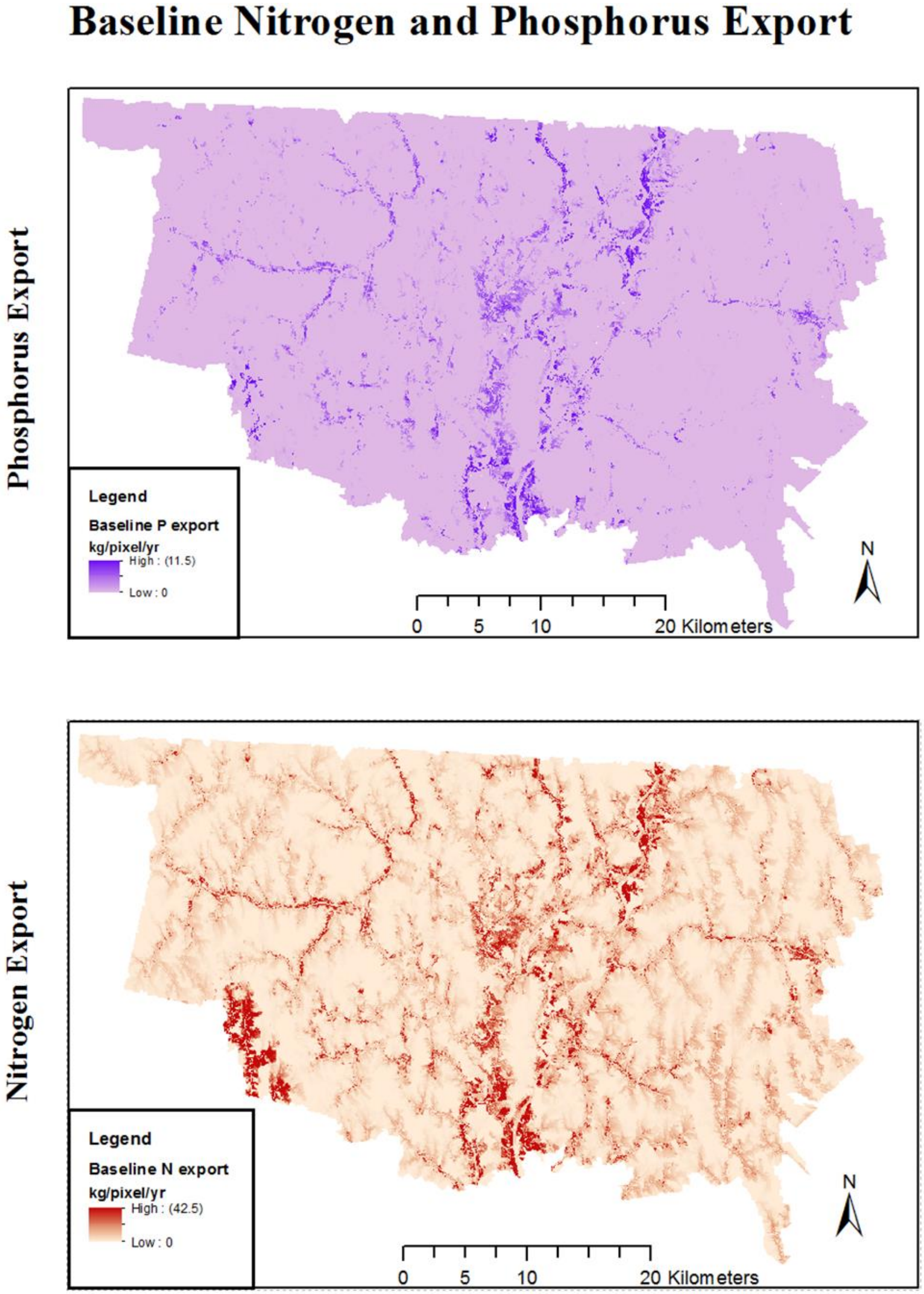


Figure 26. Map of baseline scenario nitrogen and phosphorus export. Note the geospatial distribution, in particular the heavy export in the south western portion of the county in the Westfield watershed.

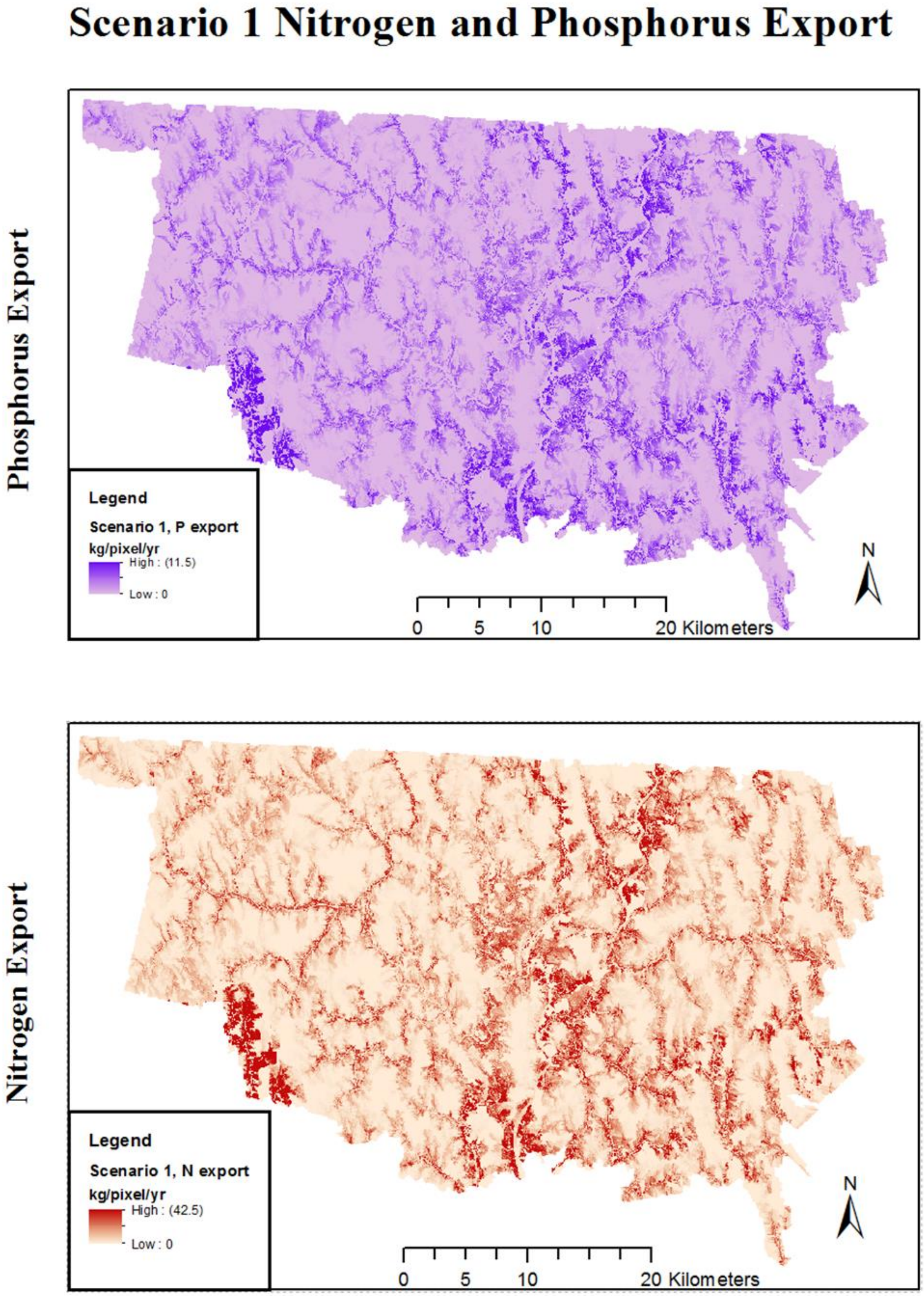


Figure 27. Map of scenario 1 nitrogen and phosphorus export.

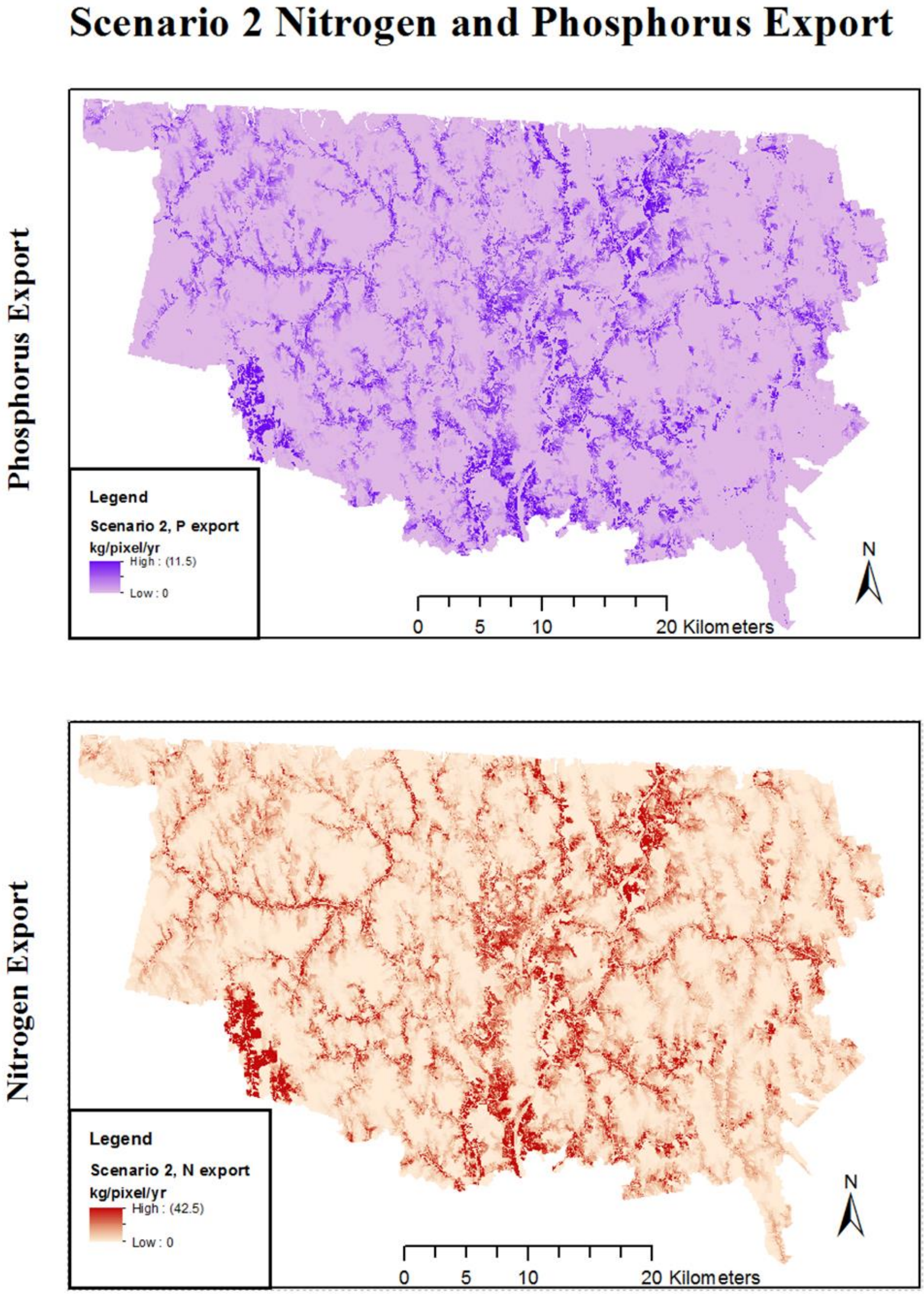


Figure 28. Map of scenario 2 nitrogen and phosphorus export.

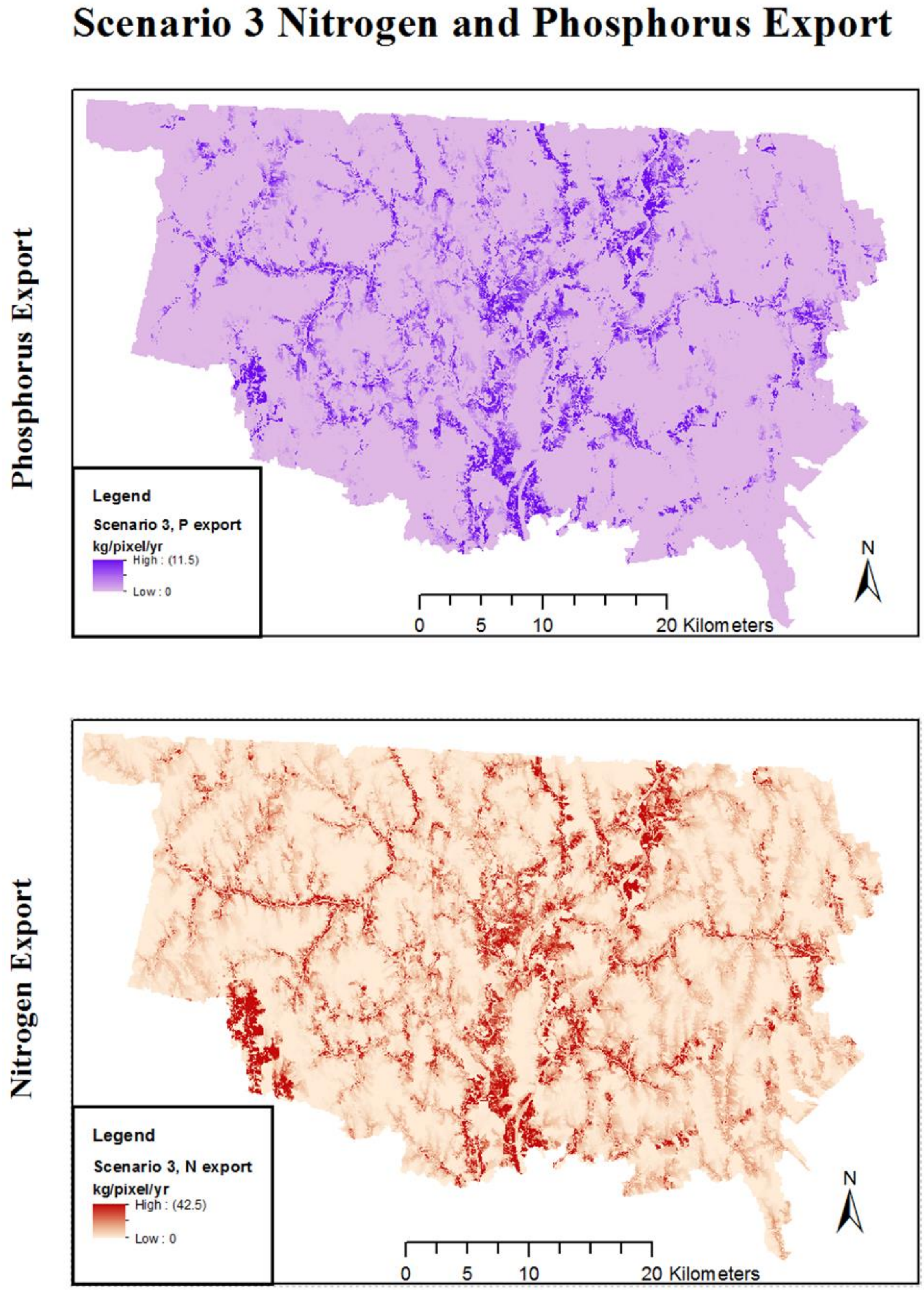


Figure 29. Map of scenario 3 nitrogen and phosphorus export.

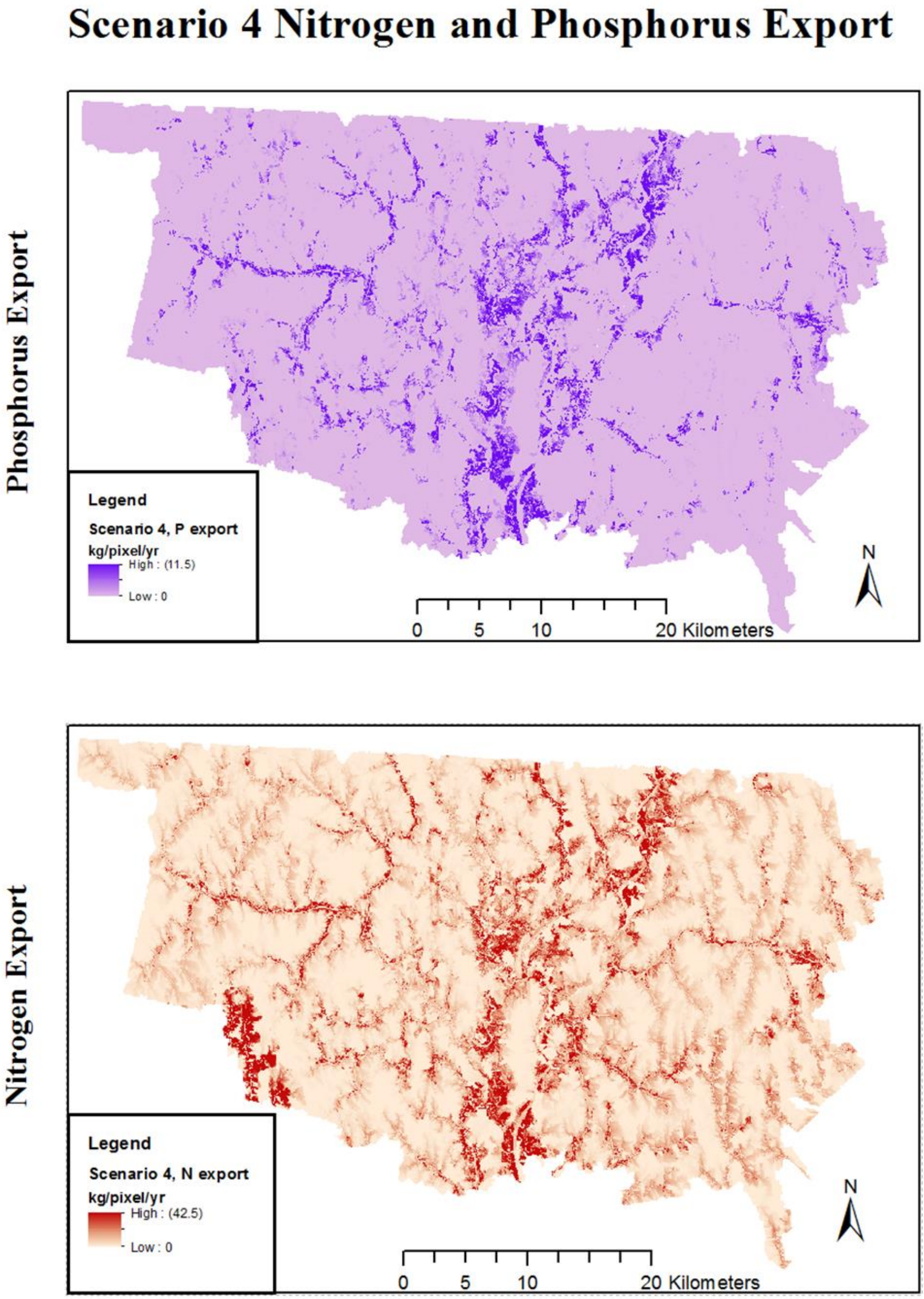


Figure 30. Map of scenario 4 nitrogen and phosphorus export.

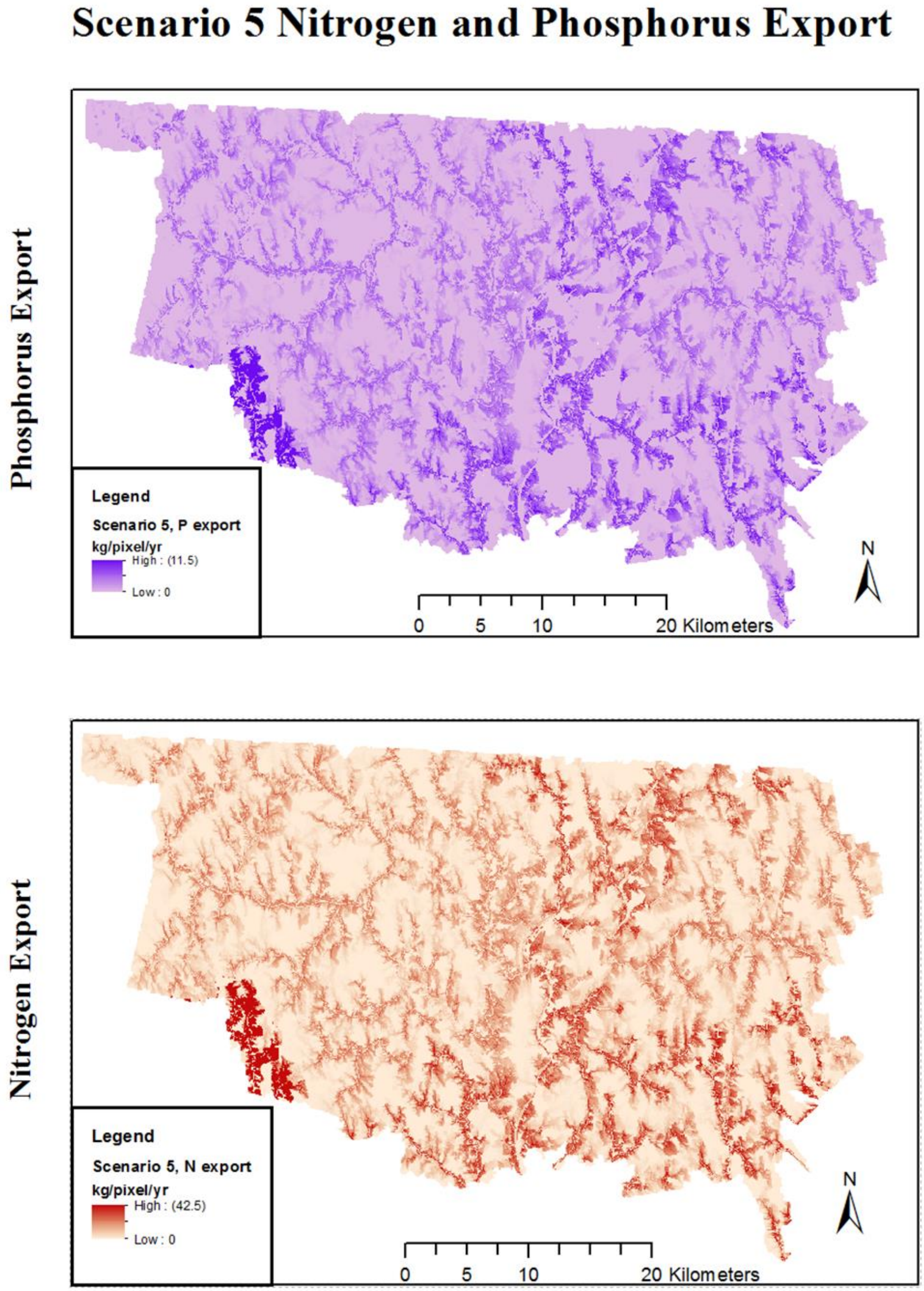


Figure 31. Map of scenario 5 nitrogen and phosphorus export.

Appendix 2

Change in Carbon Sequestration Rates under Different Scenarios

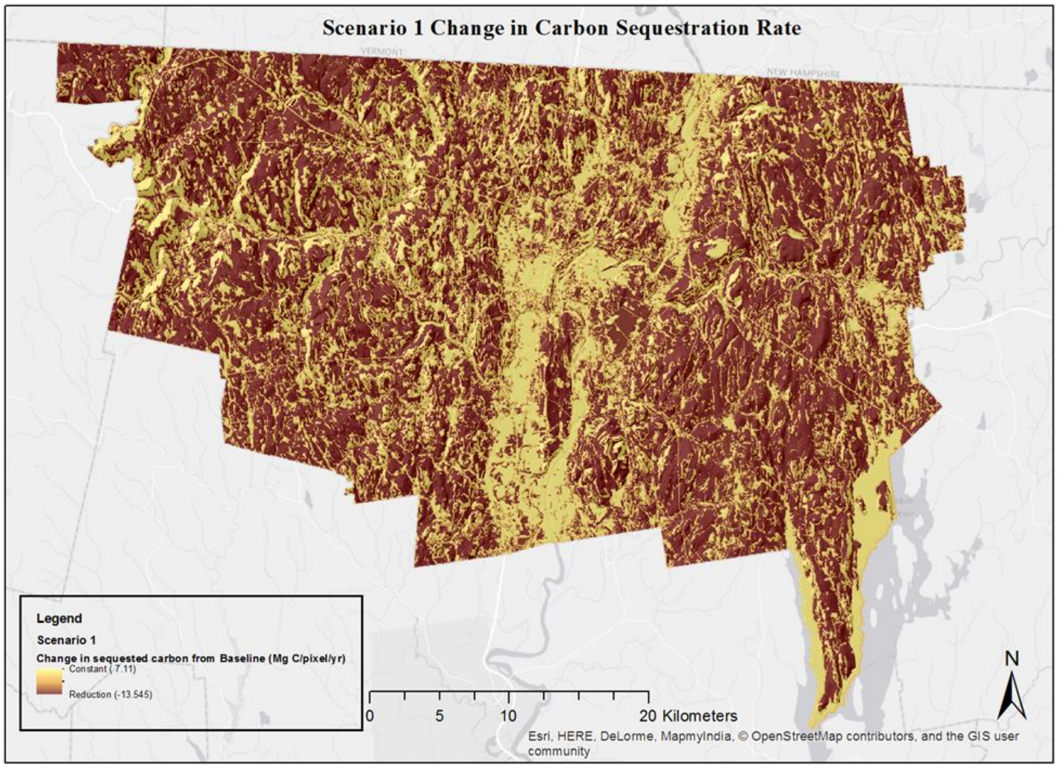


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Figure 32. Map of scenario 1 carbon sequestration rate change.

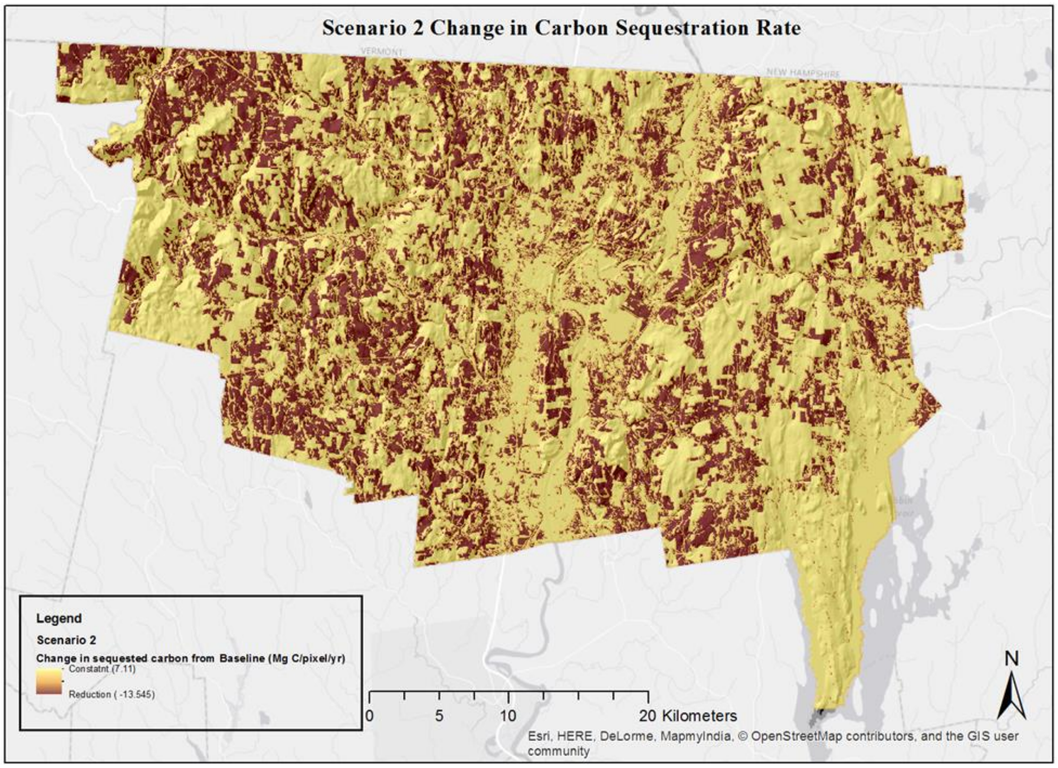


Figure 33. Map of scenario 2 carbon sequestration rate change.

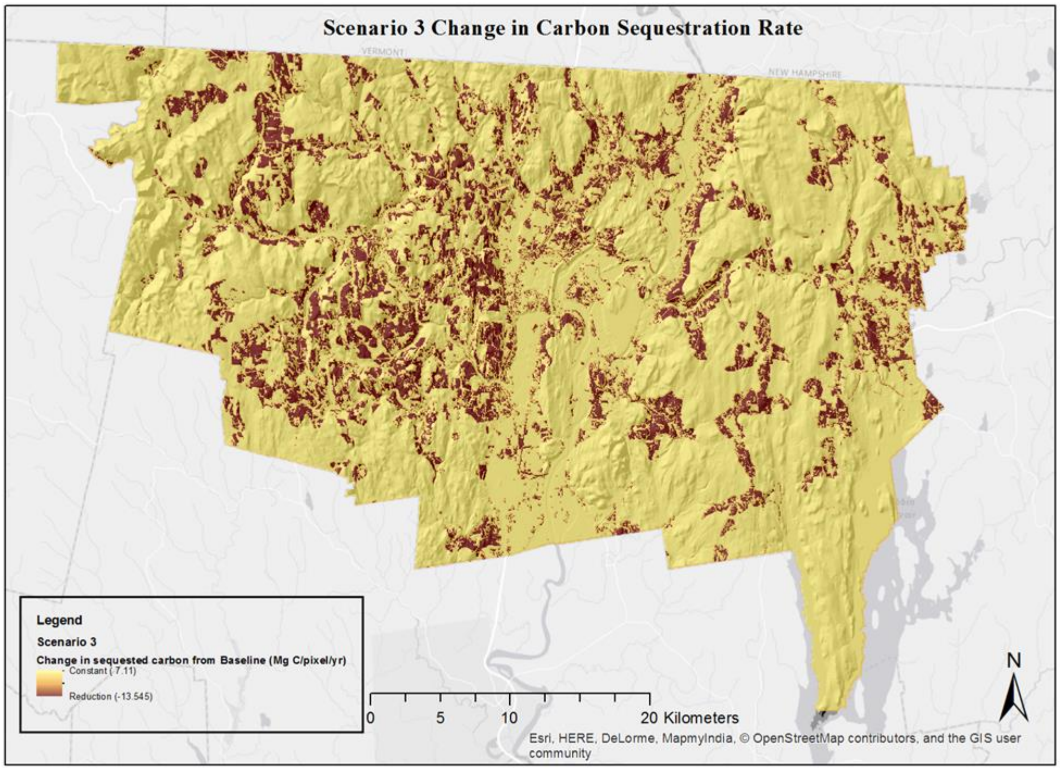
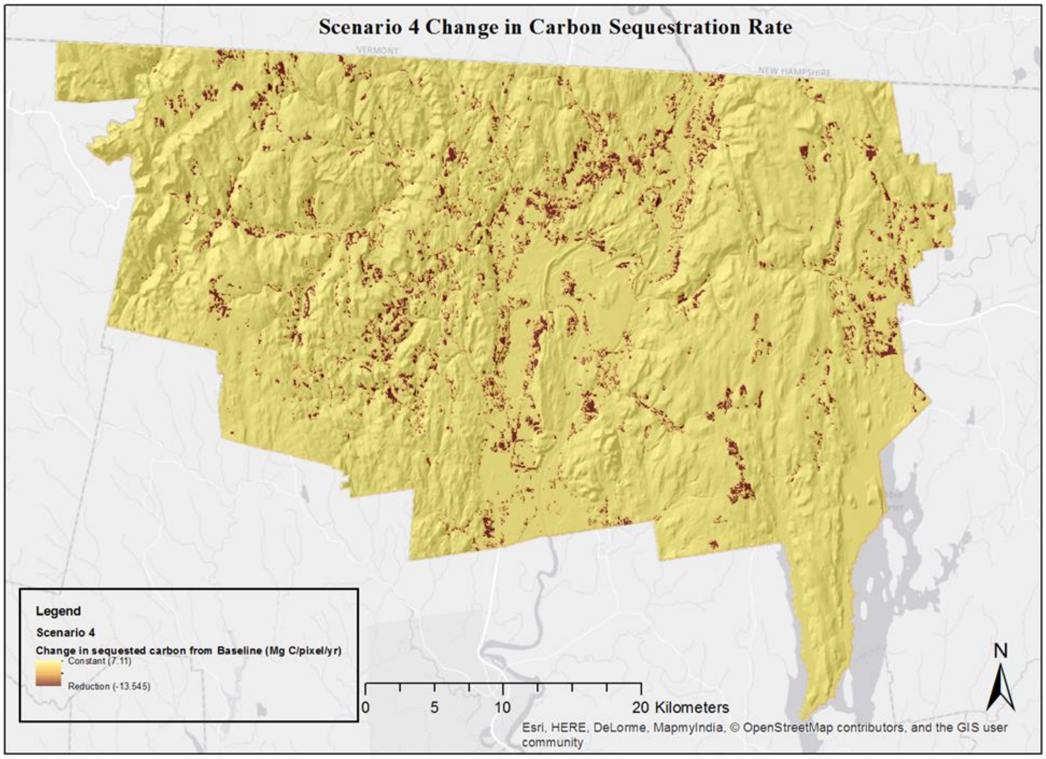


Figure 34. Map of scenario 3 carbon sequestration rate change.



Scenario 4 Change in Carbon Sequestration Rate

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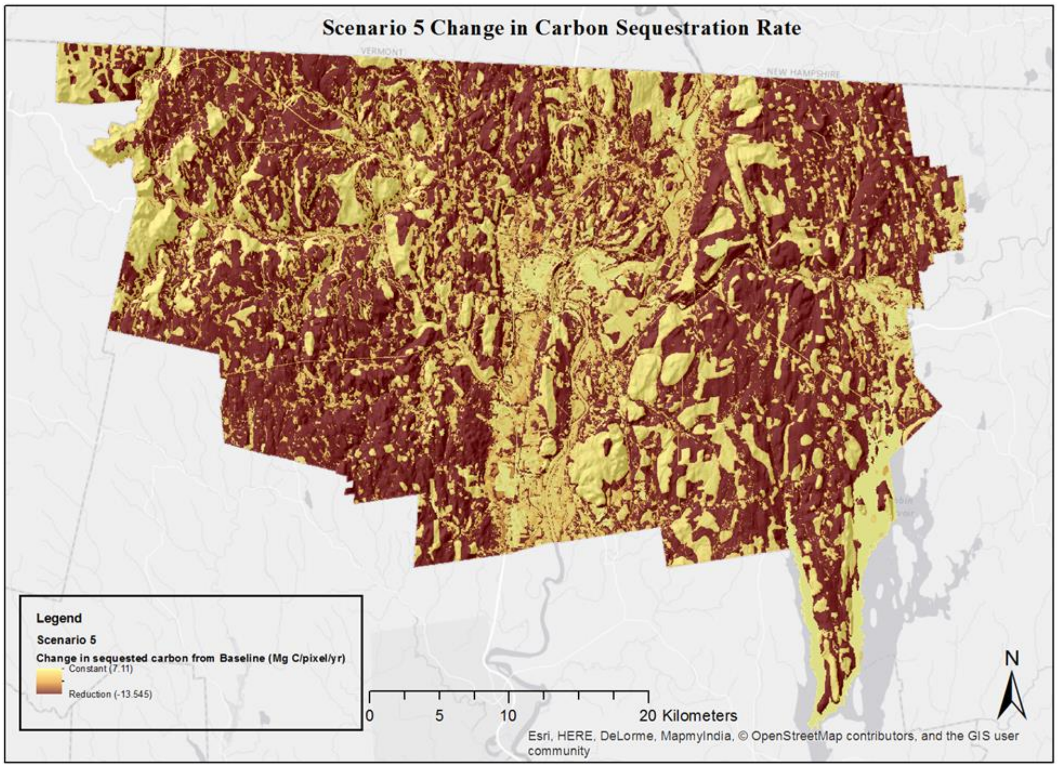
Figure 35. Map of scenario 4 carbon sequestration rate change.



Scenario 5 Change in Carbon Sequestration Rate

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Figure 36. Map of scenario 5 carbon sequestration rate change.

References

Ackerman-Leist, P. (2013). *Rebuilding the foodshed : how to create local, sustainable, and secure food systems*. Santa Rosa, CA; White River Junction, VT: Post Carbon Institute, Chelsea Green Pub.

Blayney, D., & Normile, M. A. (2004). *Economic effects of US dairy policy and alternative approaches to milk pricing.* Paper presented at the US Department of Agriculture, Report to Congress, July.

Blumstein, M., & Thompson, J. R. (2015). Land-use impacts on the quantity and configuration of ecosystem service provisioning in Massachusetts, USA. *Journal of Applied Ecology*.

Bommarco, R., Kleijn, D., & Potts, S. G. (2013). Ecological intensification: harnessing ecosystem services for food security. *Trends in ecology & evolution, 28*(4), 230-238.

Bowell, B., Coffin, C., Courchesne, C., Frignoca, I., Greene, M., Iarrapino, A., . . . Ruhf, K. (2014). New England Food Policy: Building a Sustainable Food System: American Farmland Trust; Conservation Law Foundation; Northeast Sustainable Agriculture Working Group.

Brauman, K. A., Daily, G. C., Duarte, T. K. e., & Mooney, H. A. (2007). The nature and value of ecosystem services: an overview highlighting hydrologic services. *Annu. Rev. Environ. Resour., 32*, 67-98.

Bridgham, S. D., Megonigal, J. P., Keller, J. K., Bliss, N. B., & Trettin, C. (2006). The carbon balance of North American wetlands. *Wetlands, 26*(4), 889-916.

Brown, L. R. (2004). *Outgrowing the earth: the food security challenge in an age of falling water tables and rising temperatures*: WW Norton & Company.

Brown, L. R. (2011). *World on the edge: how to prevent environmental and economic collapse*: WW Norton & Company.

Buchholz, T., Friedland, A. J., Hornig, C. E., Keeton, W. S., Zanchi, G., & Nunery, J. (2014). Mineral soil carbon fluxes in forests and implications for carbon balance assessments. *GCB Bioenergy, 6*(4), 305-311.

Canadell, J., Mooney, H., Baldocchi, D., Berry, J., Ehleringer, J., Field, C., . . . Jackson, R. B. (2000). Commentary: Carbon metabolism of the terrestrial biosphere: A multitechnique approach for improved understanding. *Ecosystems, 3*(2), 115-130.

Ciais, P., Sabine, C., Bala, G., Bopp, L., Brovkin, V., Canadell, J., . . . Midgley, P. M. (2013). Carbon and Other Biogeochemical Cycles. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom; New York, NY, USA.: Intergovernmental Panel on Climate Change.

Colby, S. L. O., Jennifer M. (2015). *Projections of the size and composition of the U.S. Population: 2014 to 20260*. United States Census Bureau.

Commission, C. B. (2004). *Cost-effective strategies for the bay: Smart investments for nutrient and sediment reduction*: Chesapeake Bay Commission.

Commonwealth of Massachusetts, D. T. N. C. (2010). *BioMap2 Conserving the Biodiversity of Massachusetts in a Changing World*. Retrieved from http://www.mass.gov/anf/research-and-tech/it-serv-and-support/application-serv/office-of-geographic-information-massgis/datalayers/biomap2.html

Compton, J. E., & Boone, R. D. (2000). Long-term impacts of agriculture on soil carbon and nitrogen in New England forests. *Ecology, 81*(8), 2314-2330.

Cronon, W. (2011). *Changes in the land: Indians, colonists, and the ecology of New England*: Macmillan.

Day, G. M. (1953). The Indian as an ecological factor in the northeastern forest. *Ecology, 34*(2), 329-346.

Diop, T. L. P. H. A. (2008). *The Organic Green Revolution*.

Donahue, B. (2007). Another Look from Sanderson's Farm: A Perspective on New England Environmental History and Conservation. *Environmental History, 12*(1), 9-35. doi:10.1093/envhis/12.1.9

Donahue, B., Burke, J., Anderson, M., Beal, A., Kelly, T., Lapping, M., . . . Berlin, L. (2014). A New England food vision. *Food Solutions New England, University of New Hampshire http:// www. foodsolutionsne. org/ sites/ default/ files/ LowResNEFV_ 0. pdf. Accessed, 10*. Retrieved from http://www.foodsolutionsne.org/sites/default/files/LowResNEFV_0.pdf

Edwards-Jones, G., i Canals, L. M., Hounsome, N., Truninger, M., Koerber, G., Hounsome, B., Plassmann, K. (2008). Testing the assertion that ‘local food is best’: the challenges of an evidence-based approach. *Trends in Food Science & Technology, 19*(5), 265-274.

Eggleston, S. (2006). *Estimation of Emissions from CO2 Capture and Storage: the 2006 IPCC Guidelines for National Greenhouse Gas Inventories.* Paper presented at the Presentation at the UNFCCC workshop on carbon dioxide capture and storage.

Ehrlich, P. (1970). The population bomb. *New York Times*, 47.
Ehrlich, P. R., & Ehrlich, A. H. (2009). The population bomb revisited. *The electronic journal of sustainable development, 1*(3), 63-71.

EPA, U. S. (2015). *A Compilation of Cost Data with the Associated Impacts and Control of Nutrient Pollution*. (EPA 820-F-15-096). U.S. EPA Office of Water.

FAO. (2009). *Food and Agriculture Organization of the United Nations: How to Feed the World in 2050*. Retrieved from Rome: http://www.fao.org/fileadmin/templates/wsfs/docs/expert_paper/How_to_Feed_the_World_in_2050.pdf

Foster, D. (2006). *Forests in time: the environmental consequences of 1,000 years of change in New England*: Yale University Press.

Foster, D., Aber, J., Cogbill, C., Hart, C., Colburn, E., D'Amato, A., . . . Fahey, T. (2010). Wildlands and woodlands: A vision for the New England landscape.

Foster, D. R., Donahue, B. M., Kittredge, D. B., Lambert, K. F., Hunter, M. L., Hall, B. R., . . . Hart, C. M. (2010). *Wildlands and Woodlands: A Vision for the New England Landscape*. Retrieved from Cambridge Massachusetts: http://harvardforest.fas.harvard.edu/news/wildlands-and-woodlands-vision-new-england-landscape

Gillman, M. W., & Ludwig, D. S. (2013). How early should obesity prevention start? *New England Journal of Medicine, 369*(23), 2173-2175.

Hall, B. M., G.; Foster, D. R.; Syfert, M.; Burk, J. . (2002). *1830 Map Project: Three hundred years of forest and land-use change in Massachusetts* [GIS Data Layer]. Retrieved from: http://harvardforest.fas.harvard.edu/hf122-detailed-metadata#sthash.7RmafYm2.dpuf

Hardin, G. (1968). The tragedy of the commons. *Science, 162*(3859), 1243-1248. doi:10.1126/science.162.3859.1243

Harding Jr, L., Gallegos, C., Perry, E., Miller, W., Adolf, J., Mallonee, M., & Paerl, H. W. (2015). Long-Term Trends of Nutrients and Phytoplankton in Chesapeake Bay. *Estuaries and Coasts*, 1-18.

Harvard University, T. H. C. S. o. P. H. Healthy Eating Plate. Retrieved from http://www.hsph.harvard.edu/nutritionsource/

Holland, R. L., Dwight; Ferguson, Kevin. (2014). HOW MUCH MEAT TO EXPECT FROM A BEEF CARCASS. In U. O. T. I. o. Agriculture (Ed.).

Houle, J. J., Roseen, R. M., Ballestero, T. P., Puls, T. A., & Sherrard Jr, J. (2013). Comparison of maintenance cost, labor demands, and system performance for LID and conventional stormwater management. *Journal of environmental engineering, 139*(7), 932-938.

Keough, G. R. (2012). *New England Fruits and Vegetables, 2012 Crop*. Retrieved from Concord, NH: http://www.nass.usda.gov/Statistics_by_State/New_England_includes/Publications/05frtveg.pdf

Lal, R. (2004). Soil carbon sequestration impacts on global climate change and food security. *Science, 304*(5677), 1623-1627.

Lane, L. E. E. (2012). *Franklin County Farmland & Foodshed Study*. Retrieved from http://issuu.com/conwaydesign/docs/franklincounty20120522_hires?e=1127520/3620336

Lin, J. P. (2004). *Review of published export coefficient and event mean concentration (EMC) data*.

MassGIS. (2015). MassGIS Datalayers. Retrieved from http://www.mass.gov/anf/research-and-tech/it-serv-and-support/application-serv/office-of-geographic-information-massgis/datalayers/layerlist.html

Mattson, M. D., Barletta, R. A., Godfrey, P. J., Wagner, K. J., & Aiello, A. (2003). Eutrophication and aquatic plant management in Massachusetts: Final generic environmental impact report.

McCullough, D. E. (2013). Hypoxia in the Northern Gulf of Mexico: A Literature Review of Causes, Impacts, and Management of Nutrient Loading in the Mississippi River-Atchafalaya River Basin (MARB) Watershed.

McFarlane, K. J., Torn, M. S., Hanson, P. J., Porras, R. C., Swanston, C. W., Callaham Jr, M. A., & Guilderson, T. P. (2013). Comparison of soil organic matter dynamics at five temperate deciduous forests with physical fractionation and radiocarbon measurements. *Biogeochemistry, 112*(1-3), 457-476.

Moorman, M. C., Hoos, A. B., Bricker, S. B., Moore, R. B., García, A. M., & Ator, S. W. (2014). *Nutrient load summaries for major lakes and estuaries of the Eastern United States, 2002* (2327-638X).

Myers, S. S., & Bernstein, A. (2011). The coming health crisis. *Scientist, 25*, 32-37.

NASS, U. (2016). *Agricultural Prices, January 2016*. Retrieved from http://usda.mannlib.cornell.edu/MannUsda/viewDocumentInfo.do?documentID=1002

NCASI, U. F. S. (2016). *The Carbon Online Estimator; COLE 1605(b) Report for Massachusetts*. Retrieved from Durham, New Hampshire USA: http://www.ncasi2.org/COLE/datafiles/reports/COLE_Jan_19_949442.pdf

NRCS, U. (2014). *National Soil Survey Handbook (NSSH)*. USDA Retrieved from http://soils.usda.gov/technical/handbook/

Parker, J. (2011). The 9 billion-people question: Spcial Report Feeding the World. *The Economist*.

Pimentel, D. W., Anne. (2004). World Population Agriculture and Malnutrition. In W. Institute (Ed.).

Pollan, M. (2006). *The omnivore's dilemma : a natural history of four meals*. New York: Penguin Press.

Popkin, B. M. (2004). The Nutrition Transition: An Overview of World Patterns of Change. *Nutrition Reviews, 62*(7), 140-143. doi:10.1301/nr.2004.jul.S140-S143

Porter, J. R., Xie, L., Challinor, A. J., Cochrane, K., Howden, S. M., Iqbal, M. M., . . . White, L. L. (2014). *Food security and food production systems. In: Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, Chapter 7*. Cambridge, United Kingdom and New York, NY, USA Retrieved from http://www.ipcc.ch/report/ar5/wg2/

Post, W. M., & Kwon, K. C. (2000). Soil carbon sequestration and land-use change: processes and potential. *Global Change Biology, 6*(3), 317-327.

Raciti, S. M., Groffman, P. M., Jenkins, J. C., Pouyat, R. V., Fahey, T. J., Pickett, S. T., & Cadenasso, M. L. (2011). Accumulation of carbon and nitrogen in residential soils with different land-use histories. *Ecosystems, 14*(2), 287-297

Rao, P., Hutyra, L. R., Raciti, S. M., & Finzi, A. C. (2013). Field and remotely sensed measures of soil and vegetation carbon and nitrogen across an urbanization gradient in the Boston metropolitan area. *Urban Ecosystems, 16*(3), 593-616.

Ray, D. K., Mueller, N. D., West, P. C., & Foley, J. A. (2013). Yield Trends Are Insufficient to Double Global Crop Production by 2050. *PLoS ONE, 8*(6), e66428. doi:10.1371/journal.pone.0066428

Reckhow, K. H., Beaulac, M. N., & Simpson, J. T. (1980). Modeling phosphorus loading and lake response under uncertainty: A manual and compilation of export coefficients

Renski, H. S., Susan; Hodge, Daniel; Proulx, William; Paik, Katherine; Herter, Steffen (2015). *Long-term population projections for Massachusetts Regions and municipalities*. Retrieved from University of Massachusetts Amherst: http://pep.donahue-institute.org/downloads/2015/new/UMDI_LongTermPopulationProjectionsReport_SECTION_2.pdf

Rinehart, L. (2006). Pasture, rangeland and grazing management. *National Sustainable Agricultural Information Service. ATTRA Publication# IP306*.

Rodríguez, J. P., Beard, T. D., Bennett, E. M., Cumming, G. S., Cork, S. J., Agard, J., . . . Peterson, G. D. (2006). Trade-offs across space, time, and ecosystem services. *Ecology and Society, 11*(1), 28.

Rotz, C. A., Roth, G. W., & Stout, W. L. (2002). Economic and environmental implications of small grain production and use on Pennsylvania dairy farms. *Applied engineering in agriculture, 18*(4), 417.

Russell, E. W. (1983). Indian-set fires in the forests of the northeastern United States. *Ecology*, 78-88.

Scavia, D. (2015). Chesapeake Bay Hypoxic Volume Forecasts Donald Scavia, Isabella Bertani and Mary Anne Evans 2.

Schoenian, S. (2012, 1/17/2012). Sheep 201: A beginner's Guide to Raising Sheep. Retrieved from http://www.sheep101.info/201/index.html

Sekar, I., & Randhir, T. O. (2014). Spatial Risk Assessment of Farming on Wetland Habitats in Watershed System. *Water, Air, & Soil Pollution, 225*(9), 1-13.

Seppelt, R., Fath, B., Burkhard, B., Fisher, J. L., Grêt-Regamey, A., Lautenbach, S., . . . Verburg, P. H. (2012). Form follows function? Proposing a blueprint for ecosystem service assessments based on reviews and case studies. *Ecological Indicators, 21*, 145-154.

Sharp, R., Tallis, H.T., Ricketts, T., Guerry, A.D., Wood, S.A., Chaplin-Kramer, R., Nelson, E., Ennaanay, D., Wolny, S., Olwero, N., Vigerstol, K., Pennington, D., Mendoza, G., Aukema, J., Foster, J., Forrest, J., Cameron, D., Arkema, K., Lonsdorf, E., Kennedy, C., Verutes, G., Kim, C.K., Guannel, G., Papenfus, M., Toft, J., Marsik, M., Bernhardt, J., Griffin, R., Glowinski, K., Chaumont, N., Perelman, A., Lacayo, M. Mandle, L., Hamel, P., Vogl, A.L., Rogers, L., and Bierbower, W. (2015). InVEST +VERSION+ User's Guide.: The Natural Capital

Project, Stanford University, University of Minnesota, The Nature Conservancy, and World Wildlife Fund. Retrieved from http://data.naturalcapitalproject.org/nightly-build/invest-users-guide/html/

Sloan, P., Praus, M., & Dechiara, M. (2015). *Franklin County Farm & Food System Project: A food system planning project for Franklin County linked to the New England Food Vision*. Retrieved from Greenfield, MA: http://frcog.org/wp-content/uploads/2015/10/FRCOG-FC-Farm-and-Food-System-Project-Final-Report-093015.pdf

Smoyer-Tomic, K. E., Spence, J. C., Raine, K. D., Amrhein, C., Cameron, N., Yasenovskiy, V., . . . Healy, J. (2008). The association between neighborhood socioeconomic status and exposure to supermarkets and fast food outlets. *Health Place, 14*(4), 740-754. doi:10.1016/j.healthplace.2007.12.001

Steinfield, H., Gerber, P., Wassenaar, T., Castel, V., & De Haan, C. (2006). *Livestock's long shadow: environmental issues and options*: Food & Agriculture Org.

Tang, G., & Beckage, B. (2010). Projecting the distribution of forests in New England in response to climate change. *Diversity and Distributions, 16*(1), 144-158.

Thompson, J. L., Kathy Fallon; Foster, David; Blumstein, Meghan; , & Broadbent, E. Z., Angelica Almeyda. (2014). *Changes to the Land: Four Scenarios for the Future of the Massachusetts Landscape*. Retrieved from Petersham, Massachusetts: http://harvardforest.fas.harvard.edu/sites/harvardforest.fas.harvard.edu/files/Changes%20to%20the%20Land%20-%20final%20report%20-%20Jan%202014.pdf

Thompson, J. R., Foster, D. R., Scheller, R., & Kittredge, D. (2011). The influence of land use and climate change on forest biomass and composition in Massachusetts, USA. *Ecological applications, 21*(7), 2425-2444. Retrieved from http://www.jstor.org/stable/41416668

Tomasso, L. P., & Leighton, M. (2014). The Impact of Land Use Change for Greenhouse Gas Inventories and State-Level Climate Mediation Policy: A GIS Methodology Applied to Connecticut. *Journal of Environmental Protection, 5*(17), 1572.

USDA. (1967). *Soil Survey: Franklin County Massachusetts*. Retrieved from http://www.nrcs.usda.gov/Internet/FSE_MANUSCRIPTS/massachusetts/franklinMA1967/Franklin_MA.pdf

USDA. (2016). USDA NASS Quick Stats: National Agriculture Census 2002, 2007, 2012. Retrieved from http://www.agcensus.usda.gov/Publications/2012/Online_Resources/Ag_Census_Web_Maps/

USDA, A. (2016). *USDA Weekly National Lamb Market Summary: Livestock, Meat, Wool, Pelt, & International Markets*. Retrieved from https://www.ams.usda.gov/mnreports/lswlamb.pdf

Van Fleet, T. C., Paul; Kittredge, David. Chapter 61 Programs: Understanding the Massachusetts Ch.61 Current Use Tax Programs. In M. D. o. C. a. Recreation (Ed.).

Verburg, R., Stehfest, E., Woltjer, G., & Eickhout, B. (2009). The effect of agricultural trade liberalisation on land-use related greenhouse gas emissions. *Global Environmental Change Part A: Human & Policy Dimensions, 19*(4), 434-446. doi:10.1016/j.gloenvcha.2009.06.004

Vihervaara, P., Rönkä, M., & Walls, M. (2010). Trends in ecosystem service research: early steps and current drivers. *Ambio, 39*(4), 314-324.