Forest Carbon Storage in the Northeastern United States: Effects of Harvesting Frequency and Intensity Including Wood Products

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FOREST CARBON STORAGE IN THE NORTHEASTERN UNITED STATES: EFFECTS OF HARVESTING FREQUENCY AND INTENSITY INCLUDING WOOD PRODUCTS

A Thesis Presented

by

Jared S. Nunery

to

The Faculty of the Graduate College

of

The University of Vermont

In Partial Fulfillment of the Requirements for the Degree of Master of Science Specializing in Natural Resources

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ABSTRACT

Temperate forests are an important carbon sink, yet there is debate regarding the net effect of forest management practices on carbon storage. Few studies have investigated the effects of different silvicultural systems, and the relative strength of in-situ forest carbon versus wood products pools remains in question. Our research (1) describes the impact of harvesting frequency and degree of post-harvest structural retention on carbon storage in northern hardwood-conifer forests, and (2) tests the significance of including harvested wood products in carbon accounting at the stand scale. We stratified Forest Inventory and Analysis (FIA) plots to control for environmental, forest structural and compositional variables, resulting in 32 FIA plots distributed throughout the northeastern U.S. We used the USDA Forest Vegetation Simulator to project stand development over a 160 year period under nine different forest management scenarios. Simulated treatments represented a gradient of increasing structural retention and decreasing harvesting frequencies and included a “no harvest” scenario. The simulations incorporated carbon flux between aboveground forest biomass (dead and live pools) and harvested wood products (including carbon storage in landfills). Mean carbon storage over the simulation period, including carbon stored in harvested wood products, was calculated for each silvicultural scenario. We investigated tradeoffs among scenarios using a factorial treatment design and two-way ANOVA. The predictive strength of management scenarios relative to site-specific variables was evaluated using Classification and Regression Trees. Mean carbon sequestration was significantly (a = 0.05) greater for “no management” compared to any of the active management scenarios. Of the harvest treatments, those favoring high levels of structural retention and decreased harvesting frequency stored the greatest amounts of carbon. In order to isolate the effect of in-situ forest carbon storage and harvested wood products, we did not include the emissions benefits associated with substituting wood fiber for other construction materials or energy sources. Modeling results from this study show that harvesting frequency and structural retention significantly affect mean carbon storage. Our results illustrate the importance of both post-harvest forest structure and harvesting frequency in carbon storage, and are valuable to land owners interested in managing forests for carbon sequestration.
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CHAPTER 1: FORESTS AND CARBON: ANTHROPOGENIC IMPACTS ON FOREST CARBON CYCLING

1.1. Introduction

The strength of the terrestrial carbon sink relative to other carbon pools has been heavily studied for over three decades. Significant effort has been put forth in quantifying the anthropogenic impacts on the terrestrial carbon sink. More specifically, studies conducted over the last two decades have shown that forest management has a significant effect on carbon storage in a variety of pools within forest ecosystems. Recent emphasis on the function of forest ecosystems in climate change mitigation has highlighted a need for a more comprehensive understanding of the effects of forest management on carbon sequestration, as well as, acknowledged three important challenges facing carbon offset projects (additionality, leakage, and permanence). The use of forest growth and yield models allows for the investigation of the relative impacts of forest management activities on carbon sequestration. In this thesis, we employ an empirical forest growth model to investigate the relative impacts of a spectrum of forest management techniques. Results from this research will help answer key questions related to the ability of forests in the northeastern U.S. to offset anthropogenic emissions. Specifically this thesis will help answer the question of how to prove that carbon sequestration in managed forests is additional to that which would have already been sequestered without changes in forest management (proof of additionality).
1.2 The carbon cycle and the potential impacts on global climate

1.2.1 Climate science and the carbon cycle

In order to understand the significance of the terrestrial carbon sink in the northern hemisphere, it is important to have a solid foundation of the relationship between climate system dynamics and the carbon cycle. The Earth system includes: the atmosphere, oceans, the lithosphere (solid earth), the cryosphere (frozen ice caps), the biosphere, and the complex interactions among these several component systems (Kay and Rall 2002). In the last two decades anthropogenic influences on these interactions have been recognized by a majority of the scientific communities and governments throughout the world. In his 1988 testimony to the State Energy Committee, NASA climate scientist Jim Hansen stated that he was 99% certain that the unusually warm, globally averaged temperatures for the 1980’s could not have occurred by chance, but rather were the result of the buildup of greenhouse gases (Hecht and Tirpak 1995). The anthropogenic impacts on the global climate are now widely recognized, including the slow rise of global mean temperatures (0.2° C per decade) between 1990 to 2005 (IPCC 2007). To fully comprehend the potential consequences of anthropogenic climate forcing, a fundamental understanding of climate system dynamics is necessary.

Components of climate systems are linked by flows of energy and matter. Energy flows involve the transfer of energy from one part of the climate system to another through sensible heat flux (heat which can be directly felt or sensed caused by conduction or convection) and latent heat flux (related to the evaporation and condensation of water vapor or the freezing and melting of ice) (Harvey 2000a). Similarly, mass flows involve
the transfer of mass from one part of the climate system (or reservoir) to another. For example, evapotranspiration resulting from stomatal conductance during the photosynthetic process results in a transfer of water from forests to the atmosphere.

Atmospheric energy flows are largely driven by shortwave radiation (SWR) emitted from the sun. A portion of SWR is reflected back to space by clouds and aerosols in the atmosphere. Additionally, SWR is reflected back to space as it hits the Earth’s surface (albedo). However, some of the incoming SWR is absorbed by the atmosphere and the Earth’s surface. Energy absorbed by the Earth’s surface is re-emitted as long wave radiation (LWR) primarily through sensible and latent heat fluxes. Similarly, water and carbon (which have significant impacts on the atmospheric radiative forcing) are cycled between the Earth’s systems. Water is transferred from the lithosphere, biosphere, and oceans through evaporation, and with it moves energy (though latent heat flux). It is then returned to these systems from the atmosphere in the form of precipitation. While in the atmosphere, water vapor is the primary greenhouse gas (GHG) responsible for the greenhouse effect (Harvey 2000a). The second most prolific GHG is carbon dioxide (CO₂), which has been directly correlated to anthropogenic activities, primarily since the advent of the industrial revolution, including fossil fuel combustion and deforestation.

The carbon cycle is much more complex in comparison to the water cycle, as carbon pools within the carbon cycle have more varied residence times. The amount of carbon in terrestrial biota is roughly comparable to the amount of atmospheric carbon. Carbon stored in soil and organic detritus is about twice the amount in either the atmosphere or above-ground biota (Harvey 2000a). Carbon in the ocean mixed layer,
which interacts directly with the atmosphere, is comparable to the amount of atmospheric carbon. The overwhelming majority of the total carbon in the biosphere, atmosphere, and ocean systems is stored in the deep ocean (Harvey 2000a). The residence time of carbon in each pool can vary from a matter of days in the biosphere to thousands of years in the deep ocean. As a result of drastically different residency times within pools, modeling carbon fluxes between pools is much more complex than with other greenhouse gasses (GHG) (Harvey 2000c). Accurately modeling GHG fluxes within reservoirs becomes important when trying to forecast related radiative forcing impacts and possible feedback mechanisms associated with increased GHG atmospheric concentrations.

Carbon is different from other GHGs in that it continuously cycles between a number of reservoirs (atmosphere, terrestrial plants, biota, soils, ocean water, and ocean sediments) (Figure 1). Consequently, unlike non-carbon based GHGs that can be characterized by a single time constant for the rate of removal from the atmosphere, the complex cycling processes of carbon make it difficult to model (Harvey 2000c). This is further exacerbated by potential feedbacks within the climate and carbon cycle that alter the rate of removal on anthropogenic CO$_2$ during the next few 1,000 years. The main removal process of CO$_2$ from the atmosphere is photosynthesis, in which carbon flows from the atmosphere to the biosphere, and inflow across pressure gradients to the ocean (Harvey 2000b).

As mentioned previously, the oceans store a disproportionate amount of the carbon in the active global carbon cycle. Consequently, a small decline in the proportion of carbon stored in the ocean will result in a significant increase in atmospheric carbon concentrations. Sudden changes in oceanic circulation could result in serious alterations
in atmospheric CO$_2$ concentrations through the release of large amounts of CO$_2$ stored in deep oceanic reservoirs (Harvey 2000b). The terrestrial carbon sink (storing approximately 1900 Gt C) is significantly smaller than the oceanic sink (storing approximately 38,560 Gt C) (Harvey 2000a), and has finite carbon storage capacity (Harvey 2000c). However, the terrestrial carbon sink does store a significant amount of carbon, and is directly affected by land-use changes (Houghton 1995) and forest management (Harmon et al. 1990).

### 1.2.2 The terrestrial carbon sink

Complex interactions between carbon cycling and forest dynamics have been intensively investigated. Temperate forest sink strength has been debated ever since terrestrial carbon sinks were first recognized by Woodwell et al. (1978) three decades ago. Recent research illustrates that North American temperate forests are in fact a net carbon sink (Woodbury et al. 2007). Birdsey et al. (2007) found that over the last 10 to 15 years, North American forests were a net carbon sink, sequestering 270 ± 130 million tons of carbon per year. The direct relationship between forest carbon fluxes in North America and land-use history demonstrates significant anthropogenic influence on forest carbon cycles (Houghton 1999). In temperate northeastern forests of the United States (hereafter “the Northeast”), changes in land-use during the 20th century have resulted in increased forested land cover. The increase in forested land is largely responsible for the net carbon sink in northeastern forests (Caspersen et al. 2000).

U.S. forests store approximately 152.2 million metric tons (MMt) CO$_2$e, representing approximately 2% of global terrestrial carbon stores, with an additional
8,781 MMt CO$_2$e stored in wood products and landfills (U.S. Environmental Protection Agency 2007). Houghton et al. (1999) found that during the 1980s the U.S. terrestrial forests carbon sink offset 10 to 30 percent of U.S. fossil fuel emissions. Since this time, emission levels have risen significantly, and general trends in forest growth suggest a decline in U.S. forest carbon uptake (Birdsey et al. 2006). This decrease is partially a result of aging forests (and the consequential decreased carbon uptake rates in older forests) coupled with forest disturbances resulting in the release of carbon to the atmosphere. As forests mature, the rates of biomass accumulation and consequential carbon uptake slow (Bormann and Likens 1979, Keeton et al. 2007) (Figure 2). The current Northeast forest carbon sink is a consequence of widespread forest clearing and agricultural practices in the 18$^{th}$ and 19$^{th}$ centuries and the subsequent forest recovery through secondary succession. At the height of land clearance, approximately 75% of New England forests had been converted to open lands (Foster and Aber 2004), resulting in a significant flux of carbon to the atmosphere. Following agricultural land abandonment at the turn of the 19$^{th}$ century, forests began to reclaim un-used agricultural fields, resulting in a flux of carbon from the atmosphere to the young, fast growing forests. As the forests continue to age the carbon uptake rates decline, while maintaining positive net carbon sequestration (Figure 2). Declines in growth rates of New England old-field successional forests are reflected in a decrease in the regional carbon sink strength. As forest reach later stages of stand development in temperate and boreal forests, carbon storage is the greatest, albeit at slower uptake rates (Harmon et al. 1990, Luyssaert et al. 2008).

Late successional forests are a noteworthy carbon sink globally, and are an
important carbon reservoir in the terrestrial carbon cycle. Harmon et al. (1990) demonstrated the importance of complex old-growth forests in the global carbon cycle, showing that despite comprising 0.017% of the earth’s land surface, deforestation of old-growth forests in the Pacific Northwest account for a disproportionate 2% of the total carbon released in the last 100 years. Old-growth forests act as carbon sinks, storing significant amounts of carbon in both living and dead biomass, as well as, belowground (carbon stored in both live and dead fine and coarse roots as wells as in organic and mineral soil layers). A review of the literature suggests a range of soil carbon storage values in mid to late successional forests (largely a factor of soil and forest type) with proportions as high as 50% of total forest carbon sequestration (Turner et al. 1995b). Furthermore, research has shown that soil carbon levels are not as affected by forest management practices as previously thought (Yanai et al. 2003). In order to more fully understand forest ecosystem carbon fluxes, further research is needed to better understand soil carbon dynamics, recognizing their significant contribution to total forest carbon sequestration.

1.2.3 Forests and climate change

Changing climatic conditions will inevitably have serious impacts on forests of the Northeast. Forecasting these climatic changes has proven difficult, as they are largely dependent on future emission levels and many underlying uncertainties associated with climate change. However, research has shown that mean global temperatures are rising, and will continue to rise under increased emissions levels (Figure 2Figure 3). Increased intensity of precipitation in the Northeast, primarily in the form of rain
(Frumhoff et al. 2007), coupled with decreased frequency of storms will have serious impacts on northeastern forests (Hayhoe et al. 2007). White et al. (1999) found through modeling that after 2050, the continued increase in temperature and precipitation will reach a threshold where they begin to adversely affect global forests and NPP will decline, especially in temperate forests. Increased CO$_2$ concentrations have been shown to increase water use efficiency and tree growth on sites not limited in soil nutrients, nitrogen, or water (Aber et al. 2001, Nowak et al. 2004). However, this temporary increase in tree growth in response to increased atmospheric CO$_2$ is not sustained indefinitely (Harvey 2000c). Studies investigating this “CO$_2$ fertilization” effect on plant life have found a multitude of feedback mechanisms associated with the resulting short-term increase in growth with CO$_2$ fertilization (Harvey 2000c). These impacts include several feedback mechanisms such as increased C:N ratio in forested ecosystems and increased downregulation in plants (changes in cellular components), both of which result in decreased plant growth (Harvey 2000c). In general, the complexities within ecosystem interactions are so great it is likely that feedback mechanisms will vary significantly across ecosystems. To date, nearly all forest Free Air CO$_2$ Enrichment (FACE) studies have focused primarily on small homogeneous stands to minimize experimental variables (Reich et al. 2006). For this reason, only limited predictions can be based on these studies regarding how landscape scale, heterogeneous forest systems will respond to increased CO$_2$ levels.

Moreover, variations in climatic conditions will cause changes in disturbance regimes in the Northeast (Dale et al. 2001). Natural disturbances play a critical role in stand development (Franklin et al. 2002, Seymour et al. 2002, Keeton et al. 2007) and
have significant effects on carbon storage (McNulty 2002). Changes in disturbance regimes, caused by global climate change and spread of exotic organisms, will likely impact carbon storage in forest ecosystems. The impact of these changes should be considered when addressing the issue of permanence of carbon stored in forests in relation to carbon offset programs.

Landscape fragmentation coupled with the increase of exotic plants, insects and pathogens have potentially detrimental effects on the resiliency of northeastern forests in response to climate change. Historically, individual species have responded to climate change with shifts in species ranges (Delcourt and Delcourt 1988). Today plant species assemblages, as well as individual species, face the greater challenge of responding to a rapidly changing climate in a severely fragmented landscape. Pitelka et al. (1997) noted that anthropogenic changes in the landscape may impede the retreat or advance of species range in response to climate change. Changes in regional precipitation and mean annual temperatures may potentially result in species range shifts (Beckage et al. 2008) and changes in species composition (Xu et al. 2009). Warmer temperatures at higher latitudes may facilitate the spread of exotic pathogens, as well as increase respiration rates, particularly in high latitude wetland areas where increased respiration will result in the emission of significant amounts of CH$_4$, a GHG over 20 times more potent than CO$_2$. Multiple impacts ensuing from feedback mechanisms correlated with increased GHG concentrations are already being seen in the northeastern U.S. Since 1970, the Northeast has been warming at a rate of 0.5°F per decade (winter temperatures are rising 1.3°F per decade) (Hayhoe et al. 2006). A study conducted by the Union of Concerned Scientists (2006), compared regional climate change scenarios for the Northeast under different
projected GHG emissions levels. They found that under low-emissions, annual temperatures are projected to increase 3.5 to 6.5°F by 2100, and 6.5 to 12.5 °F under high-emissions scenario. The same study showed a host of other impacts resulting from anthropogenic emissions of GHGs. Most notable was the “migrating States” theory. This predicts that by 2100, under continued high-emissions, Massachusetts will have the same climate as Maryland does currently (Hayhoe et al. 2006).

1.3 Forest stand dynamics in relation to carbon sequestration

1.3.1 Temperate forest stand development processes: A comparison of stand development models.

In the early 20th century, when ecologists first began to classify community types in relation to successional pathways, an emphasis was placed on orderly processes occurring along defined linear pathways with a terminus (Clements 1916, Egler 1952, Daubenmire 1966, Odum 1969). Though this paradigm prevailed for nearly eight decades, ecologists began to recognize the impact of dynamic processes such as gap dynamics (Pickett and White 1985) on successional development. Research exploring the correlated effects of disturbance and successional patterns reformulated the extant paradigms, recognizing that succession does not necessarily have a terminus (Connell and Slatyer 1977), but rather is a dynamic process with multiple pathways (Fastie 1995). The multiple pathways of succession described by Fastie (1995) in Glacier Bay, Alaska, challenged the previous work on succession. Fastie showed that spatial (distance from seed source) and temporal (time since disturbance) variability in site environmental
characteristics can dictate the process and pace of succession. Research continues today to better our understanding of successional processes over time and more specifically how these processes affect forested ecosystem.

Similar to our understanding of successional dynamics in ecosystems, the complexity of stand developmental models has also increased. Over the last few decades multiple stand development models have been published in the literature. Early models (Bormann and Likens 1979, Oliver and Larson 1996), compartmentalize stand development into four discrete phases. These initial models predicted that biomass accumulation within stands would reach an early steady state (~170 years) of equilibrium in stand development, driven by decreased NPP driven by declines in growth rates balanced by increased decomposition and respiration rates. One of the initial models (Bormann and Likens 1979) did recognize that stands never truly reach a state of complete equilibrium, but rather a state of shifting equilibrium driven by gap dynamics. Though useful pedagogical models of early development, these early models were restricted in applicability to even-aged stand development, such as plantations, and followed a single pathway of stand development. Furthermore, these models did not account for the impact of disturbance in early stand development, or the impact of biological legacies (large, live trees standing following a major disturbance) (Franklin et al. 2002).

More recently proposed models (Spies 1997, Franklin et al. 2002) highlight the dynamic nature of forested ecosystems and the subsequent impacts on stand development (Table 1). As noted by Spies (1997), stand development in the first two phases is well understood, hence we see similarities in the initial phases in all four models. The first
phase in all four models is the initiation of stand development. This phase follows a major disturbance that removes a majority of the forest canopy. This phase in development is marked by many pedologic changes (Oliver and Larson 1996), and rapid changes in species dominance, micro-environment, structure (Spies 1998), and level of competition (Spies 1997). It should be noted that one stand development model does address this initial stage of development slightly differently. Franklin et al. (2002) highlights the role of biological legacies following disturbance (Table 1). Biological legacies are defined as features that remain on a site following a natural disturbance. Recognizing the ecological function of biological legacies in stand development following disturbance as well as throughout later stages of stand development as legacy trees persist, the retention of legacy trees is now a critical component of disturbance based forestry and silvicultural practices that strive to emulate natural disturbance patterns. The fundamental understanding of the function of biological legacies emerged from studies following the Mount St. Helens eruption (Franklin and MacMahon 2000), and was further studied throughout the Pacific Northwest (Keeton and Franklin 2005). The critical role of legacy trees across a variety of ecosystem functions has since been widely recognized (Mazurek and Zielinski 2004, Keeton and Franklin 2005, Franklin et al. 2007). Additionally biological legacies can alter the pathways of stand development and successional pathways (Franklin et al. 2002, Keeton and Franklin 2005); hence the recognition of the influence of legacies in developmental models is vital.

Recognizing the stochastic nature of disturbance and multiple pathways of succession is critical in understanding stand developmental processes. Following the initial phase of stand development, the stand enters a period of intense competition and
rapid biomass accumulation (it should be noted that on some very low productivity sites, stands may stagnate in this stage for extended periods of time (Oliver and Larson 1996)). During these early developmental phases density-dependent mortality is high, and stands are rather homogeneous in structure.

After the initial stages of stand development, the stand development models begin to diverge. In all models, increased vertical complexity within the stand canopy marks the beginning of this phase. In more simplistic models based on even-aged stand development, this phase marks the beginning of a transition to a steady-state. In more complex models, this indicates the beginning of one of the most dynamic periods of stand development. More frequent small-scale disturbances increase density-independent (or agent-based) mortality, resulting in enhanced horizontal diversity within stands (Franklin and Van Pelt 2004). In these models, stand development continues for much greater time periods, as small-scale, high frequency disturbances interact with stand structural development. These models never reach a steady-state, but rather continue to progress in dynamic equilibrium until the next major disturbance that re-initiates stand development.

Continued adaption of developmental models is crucial as our understanding of ecological processes increases. The greatest strength of earlier models is in their simplicity, and ability to cleanly compartmentalize forest stand development, making them easily relatable to forest growth models. However, as we see in more complex models, this also proves to be their greatest weakness. The lack of inclusion of the dynamic nature of stand development decreases the ability of earlier models to accurately predict forest growth. This will become even more apparent with the inclusion of impacts of changing climatic conditions on forest stand development. Changes in
disturbance regimes, species composition, and regional climates will all alter the pathways of stand development. Ecosystem modelers forecasting impacts of climate change on forests will be limited in their ability to accurately forest growth, and generate realistic carbon values, if they rely on overly simplistic stand developmental models. Recognizing the variety of ecosystem processes associated with forest stand development only further stresses the importance of thoroughly understanding the developmental processes of forest ecosystems.

1.3.2 Impact of land-use history on stand developmental processes and carbon sequestration in the northeastern U.S.

Impacts of historic land-use practices (landscape level anthropogenic influences) in New England stretch far beyond the political boundaries of the northeastern US. Forest clearing in the 19th century impacted the global carbon cycle through alterations in the terrestrial carbon sink (Houghton 1993). The Northeast is currently a carbon sink as a result of reforestation following the widespread deforestation of the 18th and 19th century. On average, forests in the Northeast are around 60 to 90 years old, and are beginning to slow in the rate of biomass accumulation according to stand developmental models. A decrease in carbon sequestration rates correspond to a decline in the strength (not to be confused with the magnitude) of the terrestrial carbon sink. Significant amounts of carbon are currently stored in northeastern forests, despite potential declines in the rate of uptake of carbon (which is a reflection of the magnitude of the terrestrial carbon sink). Alterations in the age class distribution through forest management have been shown to affect total carbon sequestration (Cohen et al. 1996). Latter stages of forest development
store much greater amounts of carbon than younger forests (Harmon et al. 1990). For this reason we can expect to see continued carbon accumulation in northern hardwood forests as they continue to progress towards latter stages of stand development, albeit at decreasing rates of accumulation.

Related to changes in carbon dynamics, continued progress of the northern hardwood forests through stand development will have significant impacts on coarse woody debris (CWD, dead woody biomass greater than 10cm in diameter and 1 m in length on the forest floor) loading. As a result of land-use history, northeastern forests, on average, currently have relatively low volumes of CWD compared to pre-European levels. Increased structural development (including increased volumes of CWD) occurs in latter stages of stand development. CWD in the northern hardwood forests have been positively correlated to a multitude of ecosystem functions including: wildlife habitat (McKenny et al. 2006), riparian and stream system functions (Keeton et al. 2007), in-stream nitrogen dynamics (Bernhardt et al. 2003), and the in-stream retention of organic matter (Entrekin et al. 2008). Increased density-independent mortality coupled with more frequent small-scale disturbances increases CWD volumes on the forest floor and in adjacent streams. In addition, increased volumes of CWD result in increased total carbon sequestration (Jenisch and Harmon 2002). Changes in structural complexity throughout stand development result in a variety of successional dependant species. For this reason, management for a mosaic of successional stages, with connectivity provided at the landscape level, is necessary for the maintenance of biodiversity (Franklin and Lindenmayer 2009). This type of management is referred to as matrix management, and is a proven technique for landscape level management in a variety of ecosystem types.
1.4 Modeling carbon fluxes in terrestrial ecosystems

1.4.1 Why use ecological models?

Complex, non-linear biogeochemical and successional interactions of numerous processes at multiple spatial and temporal scales force ecosystem modelers to develop strict parameters to isolate areas of interest. It can be challenging to link small-scale processes with large-scale processes (Childress et al. 2002). For example, some models focus on the biogeochemical processes within forested ecosystems, while others look at interactions between species composition and successional dynamics or use a hybrid of the two approaches. Furthermore, some deterministic models are driven by physiological mechanisms, or first principles, while others are based on empirically derived relationships. Despite significant differences within model types, all models have similar roles in driving the formulation of hypotheses and in identifying uncertainties in our understanding of ecosystem dynamics. Recognizing the limitations of models, when used properly they can have powerful applications. Models serve as tools for scientists and policy makers alike, aiding in the development of our understanding of the complex interactions of forested ecosystems. In addition, models also serve as a valuable instrument for projecting changes in ecosystems as a result of various external factors, including climate change.

Within ecosystem modeling, the accuracy of predictions are constrained by the underlying assumptions in the model (Shugart 1984). A variety of model types exist,
each with strengths and weaknesses dictate the applicability of the model (Table 2). Through the use of models, researchers can address questions at very large scales, ranging from continental to global scales (Melillo et al. 1995). The Catch-22 with global scale modeling is that increased spatial scale and increased resolution are mutually exclusive; hence high-resolution is often sacrificed in order to predict interactions at large spatial scales. Once models are programmed and validated, they allow researchers to ask questions over a variety of spatial and temporal scales (depending on the defined parameters of a given model) efficiently and economically. Similar questions might take months, years, or decades to answer with empirical field-based research, whereas a model could simulate a scenario in a matter of minutes or hours. An example of the applicability of models can be seen in recent quantifications of the terrestrial carbon sink (Nightingale et al. 2007), where challenges of large scales make field-based research quantifications difficult.

Models are not always the “silver bullet” answer to predicting the future, as weaknesses embedded within models exist at both the theoretical level (limitations in model ability to represent realistic values) and in the limitations of individual models. Potential inaccuracies within individual models are compounded with increasing model complexity, or conversely with over-simplification. As observed by Pacala et al. (1993), complex models can predict observed phenomena, despite being fundamentally wrong in modeling ecosystem processes. This occurs as a result of complications with highly parameterized models, where error is compounded as the number of parameters is increased. Nightingale et al. (2007) noted that the parameters embedded in a model dictate the accuracy of the output for individual variables. This creates a paradox, where
increased number of parameters dictate not only the accuracy (here accuracy refers to the
ability to address a particular question), but also the error.

As a result of modeling limitations, modeling is not always the most appropriate
tool. Field-based research will almost always more accurately reflect realistic conditions
at smaller spatial and temporal scales. Field measurements offer insight into what is
actually happening in ecosystems, and serve as the foundational understanding that
ecosystem models are built upon. However, the temporal scale at which field research
can be applied is limited, and models can be applied over much larger time periods, as
well as forecast future conditions. Field measurements are needed as input data, and to
validate models and refine model algorithms and parameters (Potter et al. 2001).

However, validation should not be taken as a “golden ticket” for guaranteeing the
accuracy of a model. As noted by Schimel et al. (1997), the successful validation at one
site does not necessarily translate to validation across a greater spatial area. Care should
be taken whenever using a model; one should always learn the constraints of a given
model, defined by the embedded parameters and spatial and temporal limitations.

Coarse-scale models like those used in the VEMAP study (Melillo et al. 1995) are
valuable for landscape analysis, sacrificed in order to generate continental scale
predictions. Satellite-based models, models that use satellite derived coverages as
primary input data, allow scientists to address questions at the global scale. For example,
satellite-based ecosystem models can be coupled with GCMs to investigate the global
effects of climate change on net primary productivity (Cramer and Field 1999), or the
geographic distribution of major vegetation types (Melillo et al. 1995). These models, as
they use satellite derived multispectral data, have a low spatial resolution (often times
minimum grid size is greater than 8 x 8 km). The inherent large-scale of satellite-based models implying that most models must parameterize with relatively simple algorithms, based primarily on satellite-derived data (Reich et al. 1999). As the focus of these models is dictated by the resolution of multispectral input data (information from satellites), it can be difficult to accurately represent mechanisms such as nutrient dynamics, and response to regionally specific changes in environmental variables related to global climate changes at the stand scale. This can lead to an over-simplification of ecosystem processes occurring at finer scales (Jenkins et al. 1999).

Big-leaf models treat the entire canopy as one leaf, simplifying atmospheric and nutrient interactions. Big leaf models can be used at a range of scales (Jenkins et al. 1999), given the use of appropriate parameterization (Komatsu 2004). One of their greatest strengths is in their ability to project changes in forest ecosystem physiology and biochemistry in a changing climate. Parameters within Big-leaf models can easily be adjusted to address a variety of nutrient dynamics. For example, the PnET model has been used to model: forest carbon and water budgets (Aber and Federer 1992), changes in the nitrogen cycle (Aber et al. 1997), ozone effects on forest productivity (Ollinger et al. 1997), the interactive effects of increased atmospheric pollutants on forests (Ollinger et al. 2002), and the changes in photosynthetic rates at various time steps (Aber and Federer 1992, Aber et al. 1996). Despite flexibility in parameterization and accuracy in modeling the effects of changing climate, like all models, Big-leaf models have their weaknesses. One of the most significant criticisms of Big-leaf models is their lack of inclusion of species composition. As these models treat the canopy as one giant leaf, and focus on nutrient dynamics as the driving force of forest growth, they ignore the complex species
interactions and stand dynamics that have significant impacts on forest growth. Big-leaf models lack the ability to simulate changes in species diversity, community physiognomy, or changes in a species with particular significance (commercial importance) (Reynolds et al. 2001). The complex relationships between photosynthesis and nitrogen and absorbed light, coupled with the changes in leaf microenvironment with canopy height make scaling up of leaf physiology to the canopy level very difficult (Friend 2001). This has resulted in a simplification of the photosynthetic and biogeochemical processes within canopies, treating them as one “big leaf”. For this reason Friend (2001) noted the need for caution when using Big-leaf models to scale satellite estimates of leaf physiology to landscape scale forest canopies, as in the TEM model (Raich et al. 1991). Dai et al. (2004) noted that “Big-leaf” models often overestimate fluxes of CO₂ and water vapor. Adapted multilayer models have been created in an attempt to overcome the oversimplification of Big-leaf models in accurately reflecting vertical canopy fluxes and interactions. Multilayer models have increased complexity, separating the fluxes of multiple layers to obtain the total flux. However, despite their increased in modeling canopy complexities, multilayer models omit forest developmental processes, as well, as complex species interactions.

Individual tree-based models are yet another approach to modeling forest productivity. Many of these models evolved from the original JABOWA model (Botkin et al. 1972). These models consist of multiple spatial cells containing trees, where changes in the state of the forest are a function of the present state and stochastic components. The models are run using sub-models of growth, mortality, and recruitment of each tree to explore species specific stand dynamics. Unlike the process-based big-
leaf models, or the large spatial scale satellite-based models, individual tree-based models use known inter-species relationships and forest successional dynamics to project forest growth at fine spatial scales. The strengths of these models are in their ability to model forest response to structural changes in forested ecosystems, specifically changes caused by disturbance. Disturbances can be either natural or anthropogenic, though both have significant impacts on forested ecosystems that are difficult to model in satellite-based or Big-leaf models. Individual tree-based models can be either mechanistic based, as in the SORTIE model (Pacala et al. 1993), or empirically based as in the Forest Vegetation Simulator (Dixon 2002). Despite their excellent representation of forest stand dynamics, especially in response to forest management activities, these models lack the ability to accurately address forest response to climate change. The focus of individual tree-based models is not on physiologic or biogeochemical processes. Additionally, these models are often restricted to the forest stand scale. The spatial restrictions and omission of physiological mechanisms make it difficult to measure impacts of climate change, as this model type does not account the changing environmental conditions and nutrient feedbacks on growth (Reynolds et al. 2001). Although many of these models do have climatic calibrations (such as precipitation) that affect growth, climatic variables are not the driving mechanisms of plant growth.

1.4.2 Determining the appropriate model for this thesis.

This research project explores the effects of alternative forest management practices on aboveground carbon sequestration. Using an individual tree-based model allows the user to focus on inter- as well as intra-stem competition as a driving force
behind forest growth. Silvicultural practices manipulate growing space availability. The removal of one or more canopy trees is followed by a period of intense competition (Oliver and Larson 1996). Eventually vigorous individuals out-compete and suppress the growth of less vigorous individuals (Smith 1997). These rather simple stand dynamic interactions are part of the foundation from which silvicultural prescriptions have evolved. However, these interactions cannot be modeled well in either Big-leaf or satellite-based models.

The coarse resolution of satellite-based models restricts the analysis of fine-scale tradeoffs between silvicultural prescriptions, which is the focus of this study. For example, satellite-based models could investigate the changes in land-use and vegetation at the regional or continental scale; however, the differences between minute reductions in residual basal area following harvests at the stand level would be completely lost. The omission of species composition in Big-leaf models makes it difficult to reflect realistic forest management practices, as species composition is a major economic concern driving many forest management practices (Reynolds et al. 2001).

Model selection must reflect the goals of the research question, and some sacrifice in accuracy in other areas is necessary. In this study, we have chosen to forgo incorporating the potential influences of climate change on forest growth, in order to isolate the impacts of forest management on carbon sequestration. Choosing to use a Big-leaf model for the purposes of this project would not reflect the goals of the research question, as Big-leaf models are best used when addressing physiological and biogeochemical processes, where species composition and stand structure is less relevant. An individual tree-based model was used for this research, due to the spatial extent of the
research questions, as well as the model’s ability to accurately simulate forest stand dynamics, as well as incorporate species composition.

1.4.3 The Forest Vegetation Simulator

The USDA Forest Service’s Forest Vegetation Simulator (FVS) model has been used by North American forest managers for over 30 years in a variety of applications, including the development of wildlife management practices, creation of fuel reduction plans, and analysis of alternative silvicultural treatments. FVS was originally created as the Prognosis Model for Stand Development (Stage 1973), and has evolved over the last 35 years to its current state. The model accurately predicts stand and successional dynamics, as well as forest development under a range of alternative management scenarios (Crookston and Dixon 2005). FVS is a distant-independent, individual-tree forest growth model, specifically designed for applicability in both even and uneven-aged, multi-species stands (Crookston and Dixon 2005). The basic projection of FVS is the stand level; however the model also has the ability to function at the landscape level, as well as, with the incorporation of the parallel processing extension (PPE: a submodel used to project simulations of multiple stands). The temporal scope of model projections can range from 5 to several hundred years, with a 5 to 10 year resolution. The projections begin with a summary of current stand conditions based on original input data and then follow a sequential command order (Figure 4).

The FVS model contains a self-calibration feature (accuracy of calibration is dependent on level of initial stand information incorporated in original input dataset), that will modify growth rates to reflect regional characteristics. Component models (variants)
exist that are specific to regional climatic conditions, growth rates, etc. For this study the Northeast Variant of FVS (NE-FVS) was used to reflect growth rates specific to Northeast. The Northeast Variant is an adaption of the NE-TWIGS model (Hilt and Teck 1989) using the FVS framework. The variant uses growth and yield equations from NE-TWIGS, with an embedded height equation and bark ratios specific to eastern species. The Northeast Variant has several distinct differences from western variants. Mortality is based on models developed for the TWIGS family of models. Survivorship is used rather than mortality, and is predicted as a function of diameter, diameter growth, basal area in larger trees, and/or site index. The survival rate is then converted to a mortality rate when processed in FVS (Crookston and Dixon 2005). Regeneration is another major difference in variants. In some western variants, models have been developed to incorporate regeneration into FVS processing. As these models do not exist for the Northeast Variant, the user must define regeneration inputs using Event Monitor rules within FVS. Event Monitor rules can be established that define species, density, and size of expected new trees. These Event Monitors rules can then be used to emulate natural regeneration, as well as, in response to forest management. The amount of carbon sequestered in wood products is obtained from output tables, tracking carbon sinks through product (extracted forest biomass manufactured into a usable good) life history from production to landfill.

1.5 Regeneration input in model simulations

Within NE-FVS, mortality is a calculated two ways, the second of which is a
function of stocking density, reflected in the total number of trees per acre. As a result, large increases in total number of trees per acre correspond with increased mortality rates. On the other hand, low numbers of trees per acre correspond with a decreased mortality rates. Because of the critical role of trees per acre in predicting realistic forest growth values within NE-FVS, incorporating regeneration is vital, as regeneration is closely tied with total number of trees per acre.

In NE-FVS, only stump-sprouts from felled trees are automatically incorporated as regeneration in model simulations. For this reason, user defined regeneration inputs are necessary to incorporate natural regeneration and non-stump sprout regeneration as result of forest management practices. The development of the background regeneration numbers used in this study followed a two step process. First, we completed a comprehensive literature review of regeneration in northern hardwood forests, focusing both on regeneration in response to active management, as well as, natural regeneration values. Little information was available for natural regeneration values in northern hardwood forests (Graber and Leak 1992, Leak 2005). To supplement natural regeneration literature values, we used unpublished data from an ongoing study (Keeton 2006), where regeneration was measured annually from 2001-2008 in two mature northern hardwood forests in northern Vermont.

The second step required the adaption of literature and field measurement values, based on the restrictions of imbedded mortality parameters within NE-FVS (i.e., too many seedlings input into a given time-step creates unrealistic mortality as a result of the NE-FVS mortality sub-routines). To do this, we input literature and field measurement values into NE-FVS and simulated forest growth over a 100 year time
period with five different regeneration input scenarios (Figure 5). The first scenario (Regen_1) used no regeneration input. The second scenario (Regen_2) used values based on the literature input on every five year simulation cycle. The third scenario (Regen_3) used reduced literature values input on every five year simulation cycle. The fourth scenario (Regen_4) used one half of the literature values input on every other five year simulation cycle. The fifth scenario (Regen_5) used the reduced regeneration values tested in Regen_3, input on every other five year simulation cycle. In all scenarios, stump sprouts were not included.

For active management scenarios, multiple sources were available listing total number of seedlings per acre for uneven- (Mader and Nyland 1984, Leak 1987) and even-aged (Leak 1987, 2005) in northern hardwood forests. The following is a specific listing of literature values used in the development of simulation regeneration following specific management scenarios. We recognized that both the total number of regenerating seedlings and species composition of regenerating seedling varies between management scenarios. For clearcut scenarios we used regeneration data from Leak et al. (1987), who found that 5 years post-harvest there were 20,000 to 30,000 seedlings per acre, 1 to 5 feet tall regenerating in a ¼ acre patch cut in northern hardwood forests. We used regenerating species composition data from Leak et al. (1987), who found that within the same patch cuts, 2/3 of the species were generally dominated by shade intolerant (*Betula papyrifera* and *Populus tremuloides*) and intermediate tolerant species (*Betula alleghaniensis*, *Fraxinus americana*, and *Acer rubrum*), and 1/3 by shade tolerant species (*Acer saccharum*, *Fagus grandifolia*, *Tsuga canadensis*, and *Picea rubens*). For shelterwood scenarios we used data from Leak et al. (1987), who found that prior to the
removal cut in a shelterwood harvest in northern hardwood forests, there were 5,000 well-distributed seedlings per acre, 3 to 4 feet tall. For the individual tree selection systems we used data from Donoso et al. (2000) and species proportions from Leak et al. (1987) to develop species specific regeneration numbers. For each forest management scenario, we chose regeneration values that most realistically predicted forest mortality rates reflective of northern hardwood forests. Additionally, species composition of regenerated species reflected percent canopy removal of harvesting activities (i.e., increased canopy removal = increased proportion of shade intolerant species, and decreased canopy removal = increased proportion of shade tolerant species).

1.6 Managing forests for carbon sequestration

1.6.1 Forest carbon pools

Carbon is cycled through forested ecosystems as it passes through various reservoirs, or pools. Carbon is sequestered through the photosynthetic process, as plants utilize energy from the sun, in combination with gaseous carbon dioxide, to form carbohydrates. Energy stored in molecular bonds of these carbohydrates is used to fuel plant metabolism, including growth and respiration. As a result of the photosynthetic process, carbon enters forested ecosystems (in the form of CO₂) through conversion to living biomass. Interestingly, it also sometimes enters ecosystems through photosynthate exudates excreted directly to the soil, where plants have evolved mutualistic relationships with belowground biota. Through the natural mortality and decompositional processes, carbon enters the soils. This happens primarily though the decay of dead material.
Research has shown that the transitional pool of dead material acts as a critical vehicle to transport substantial amounts of carbon from the living biomass to the soils (Turner et al. 1995a). Carbon storage in soils is less well understood. The soil carbon pool is defined by the balance of carbon inputs from litterfall and rhizodeposition and the release of carbon through decomposition (Jandl et al. 2007). Disturbance of organic soils, increased temperature, changes in prescription, and a host of other climatic variables all affect soil respiration rates. Though soil carbon densities vary significantly with species composition and soil type (Jandl et al. 2007), it is generally thought that in temperate forested ecosystems, soils store 50% of the total carbon stored within the forest. The effects of forest management on carbon sequestration is not completely understood (Yanai et al. 2003). However, research has shown that minimizing the disruption of forest soils during harvests will reduce the loss of carbon from soils (Jandl et al. 2007). More research is needed to understand better the effects of active forest management on soil carbon sequestration.

When investigating the effects of active forest management, the inclusion of carbon stored in wood products is necessary. When a tree is felled a portion of its biomass remains in the forest (limbs and branches in the form of slash and/or the stump and root system below the forest floor). A fraction of the removed wood is processed into a final product with a finite life span. A portion of the original carbon from the tree is lost during the manufacturing process (this can be upwards of 60% loss during the manufacturing process for some species (Harmon et al. 1996)). However, a fraction of the tree’s carbon is sequestered in the final product for variable time periods. For this reason the inclusion of carbon stored in wood products is necessary to accurately model
1.6.2 Identifying key forest management variables effecting carbon sequestration.

As awareness of the relationship between forests and carbon sequestration increases, researchers have identified need to further quantify the correlation between forest management and carbon sequestration (Birdsey et al. 2007, Ingerson 2007, Ray et al. 2009). Over the last decade, many studies have attempted to quantify the role forests play in the terrestrial carbon budget. From this body of research several key variables that help us understand the effects of forest management on forest carbon sequestration. These variables include: harvesting frequency (rotation length or entry cycle), harvesting intensity (the residual biomass following a harvest), and the inclusion of wood products in forest carbon modeling. Current voluntary carbon markets in North America, such as the Chicago Climate Exchange (CCX) and the California Climate Action Registry (CCAR), incorporate improved forest management techniques such as reduced harvesting intensity and frequency, as a viable CO₂ emission mitigation project. However, more research is needed to better understand the interactive effects of harvesting frequency and intensity on forest carbon sequestration.

1.6.3 Quantifying the impacts of forest management on carbon sequestration

In 2007, a special task force of scientists discussing the impacts of climate change specifically recognized forest management as having the potential to make substantial contributions to national and global mitigation portfolios designed to reduce the rate of carbon dioxide (CO₂) accumulation in the atmosphere (Larsson et al. 2007). Recent
research has attempted to quantify the regional carbon flux of specific forest types; however, no studies specifically addressed the forests of the Northeast (Table 2). Furthermore, none of the previous research efforts offer specific sensitivity analysis of key variables such as harvesting frequency or post-harvest structural retention, while including carbon sequestration in wood products. Only Masera et al. (2003) and Kraxner et al. (2003) used a temporal scale that was greater than one rotation or entry cycle. However, both of these studies fail to compare their results against an unmanaged reference system. Comparison against an unmanaged reference system is crucial, as it is well documented that old-growth, unmanaged forests sequester considerable levels of carbon (Harmon et al. 1990). Masera et al. (2003) compared a variety of forest types, under varying harvesting frequencies, but they failed to incorporate coarse woody debris volumes into total biomass carbon numbers. Turner et al. (1995a) found that in the conterminous U.S., 33% of the total carbon stored in forests was stored in trees, while 10% was stored in coarse woody debris. Coarse woody debris comprises a substantial proportion of total forest carbon, and should be incorporated into modeling efforts. For a comprehensive comparison of past modeling efforts see Table 3.

1.6.4 The effect of harvesting frequency on carbon sequestration

Increased rotation lengths have been shown to increase carbon storage within forests (Harmon and Marks 2002). Kaipainen et al. (2004) used modeling to show that by increasing rotation lengths by 20 years in European forests, the carbon storage could be enhanced from 0.3 to 5.1 Mg/ha depending on forest type. Furthermore, Liski et al. (2001) found that even with the inclusion of fossil carbon emissions from timber
harvesting and manufacturing, increased rotation length sequestered more carbon in *Pinus sylvestris* and *Picea abies* stands. Extending rotation lengths allow forests to recover late-successional structure, associated with high levels of biomass and carbon storage (Harmon et al. 1990). Even when disturbance is incorporated in long term modeling of forest carbon stocks, old-growth forests retain 87% of their potential carbon stores (Krankina and Harmon 1994). In comparison, the same forest managed with 60-100 year rotation lengths with no thinning or salvage logging between harvests, will store between 25-40% of maximum potential carbon storage (Krankina and Harmon 1994). These findings attest to the substantial function that forest preserves play in regional carbon sequestration.

1.6.5 The effect of post-harvest structural retention on carbon sequestration

The effect of post harvest structural retention on carbon sequestration has been less studied than harvesting frequency. Keeton (2006) showed that decreased harvesting intensity, focused on increased post-harvest structural retention, increased post-harvest aboveground biomass. The effect of traditional silvicultural prescriptions on forest structure have been studied in the northeastern U.S. (Kenefic and Nyland 2007). Traditional silvicultural techniques can lead to decreased stand structural complexities, specifically in the accretion of dead wood in forest stands in latter stages of stand development (Crow et al. 2002). Research has shown that even within intensively managed forests, silvicultural techniques can have significant effects on structural development (Ishii et al. 2008). Recognizing that management activities alter natural stand developmental dynamics, it is intuitive that similar effects would be seen in carbon
sequestration. For this reason, more research is needed to address specifically the effects of post-harvest structural retention associated with forest management activities on carbon sequestration.

1.6.6 Harvested wood products and forest carbon sequestration

Quantifying carbon sequestration in post-harvest products has proved to be particularly controversial. Cote et al. (2002) used a net balance methodology (as opposed to the widely used life-cycle inventory analysis) to show that the inclusion of carbon stored in wood products can increase the magnitude of forest carbon sinks. However, as this study employed a net balance quantification, emissions associated with the harvesting, transport, and manufacturing of products were not included (Cote et al. 2002).

In the U.S. Pacific Northwest, several studies have examined carbon sequestration in wood products. Harmon et al. (1996) found that from 1900 to 1992 in Oregon and Washington, 23% of harvested carbon was stored in wood products. Due to significant changes in post-harvest manufacturing, increased manufacturing efficiency has decreased the loss of carbon throughout the manufacturing process. Despite technological improvements over the 92 year time period of this study, 45-60% of harvested carbon was lost during the manufacturing process (Harmon et al. 1996). A similar study showed that 42% of timber harvested in the Pacific Northwest entered long-term storage (products with lifespan greater than five years) (Harmon et al. 1990). The definition of long-term storage is a possible source of discrepancy between studies. Several studies exploring tradeoffs between different forest carbon sinks have uniformly used one half-life to model carbon residency in wood products (Thornley and Cannel 2000, Kraxner et
Harmon et al. (1996) showed that carbon residency time varied depending on the type of wood product (i.e. paper, paper board, timber, etc.), as well as the disposal location (landfill, open air, etc.). A sensitivity analysis of landfill type (open dump or enclosed landfill) showed significant differences in carbon residency times (Harmon et al. 1996). The incorporation of landfill decay rates and varying half-life periods will allow for the most accurate accounting of wood products sinks.

In addition to elucidating the role that wood products play in actively managed forest carbon sequestration, the substitutive effect (replacing energy intensive products such as concrete and steel with durable wood products) of reduced emissions associated with this carbon pool should also be recognized. The cement industry contributes 5% of global CO$_2$ emissions (International Energy Agency 1999, as cited in Szabó et al. 2006). Cement production is expected to increase approximately four-fold from 112 Mt in 1997 to 450 Mt in 2030 (Szabó et al. 2006). The substitution of wood products for cement products could significantly reduce production emissions. However, these avoided emissions are difficult to quantify, and due to high regional variability in cement use, they must be regionally weighted (Szabó et al. 2006). Further research is needed to quantify regional substitutive impacts of the use of wood products and develop more accurate life cycle analyses.

1.7 Conclusions

Increased levels of carbon dioxide as a result of anthropogenic activities are causing disruptions in the Earth’s atmosphere (IPCC 2007). Alterations in the terrestrial
carbon sink within the carbon cycle has been shown to have a significant effect on atmospheric CO$_2$ concentrations (Houghton 1995). Within the terrestrial forests sequester substantial amounts of carbon. Recognizing the anthropogenic influence on forest ecosystems through forest management, it is critical to understand carbon stocks and fluxes associated with regional silvicultural techniques.

Previous research has independently highlighted the effects of harvesting intensity and frequency on forest carbon sequestration. Uncertainties within studies exist that are largely associated with the scale at which carbon accounting is conducted (spatial, temporal, as well as at the inclusion of non-forest carbon pools) (Harmon 2001). More research is needed that focuses on the broader spectrum of silvicultural options used in the Northeast, as well as the relative effect of the inclusion of carbon stored in wood product pools. The next chapter in this thesis describes a study developed to answer these questions. Findings from this body of work will prove to be helpful to Northeast landowners interested in managing forests for carbon sequestration.
## 1.8 Tables

Table 1: Comparison of four general stand development models. Note similar components within models highlighted in grey (Adapted from Franklin et al. (2002)).

<table>
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<tr>
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<tbody>
<tr>
<td>Disturbance and legacy creation</td>
<td>Cohort establishment</td>
<td>Stand initiation</td>
<td>Establishment phase</td>
<td>Reorganization phase</td>
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<tr>
<td>0</td>
<td></td>
<td></td>
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<tr>
<td>Canopy closure</td>
<td>Stem exclusion</td>
<td>Thinning phase</td>
<td>Aggradation phase</td>
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<td>20</td>
<td></td>
<td></td>
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<tr>
<td>Biomass accumulation/competitive exclusion</td>
<td>Understory re-initiation</td>
<td>Maturation sub-phase</td>
<td>Transition phase</td>
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<td>80</td>
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<td>Maturation</td>
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<tr>
<td>Old-growth</td>
<td>Transition phase</td>
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<td>150</td>
<td>Vertical diversification</td>
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<td>Steady-state</td>
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<td>300</td>
<td>Horizontal diversification</td>
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<td>Old-growth and late transition</td>
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<td>800</td>
<td>Pioneer cohort loss</td>
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<td>Shifting-gap phase</td>
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<td>1200</td>
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Table 2: Comparisons of a variety of model types used to model forest ecosystem dynamics, note spatial and temporal scales are relative and may vary between individual models within classes. Table adapted from Reynolds et al. 2002.

<table>
<thead>
<tr>
<th>Types of models</th>
<th>Description</th>
<th>Spatial Resolution</th>
<th>Spatial Extent</th>
<th>Temporal Resolution (time step)</th>
<th>Temporal extent (length of simulation)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Global biogeographical</strong></td>
<td>Simple relations between climate variables and vegetation features used to determine equilibrium response of vegetation to climate change</td>
<td>Coarse (0.5° x 0.5° lat/long)</td>
<td>Global</td>
<td>Static (climate-vegetation equilibrium)</td>
<td>Static (climate-vegetation equilibrium)</td>
</tr>
<tr>
<td><strong>Dynamic global vegetation</strong></td>
<td>Ecosystem fluxes of C and nutrients described as functions, representing the performances of plant canopies - e.g. water and energy fluxes and responses to elevated CO2 levels - based on simplified physiology and biophysics</td>
<td>Coarse (0.5° x 0.5° lat/long)</td>
<td>Global</td>
<td>Week-year</td>
<td>Decades-century</td>
</tr>
<tr>
<td><strong>Big-leaf models</strong></td>
<td>Forest dynamics described by simplifying the effects of environmental conditions on the growth of species; treating photosynthetic and nutrient fluxes at the canopy level as one leaf</td>
<td>Fine 1-10 ha</td>
<td>10-1000 ha</td>
<td>day-Year</td>
<td>Decades-century</td>
</tr>
<tr>
<td><strong>Multilayer models</strong></td>
<td>Similar to Big-leaf in that they are driven by photosynthetic and nutrient fluxes; however these models incorporate multiple vertical layers of the forest canopy to explore complex vertically stratified interactions</td>
<td>Fine 1-10 ha</td>
<td>10-1000 ha</td>
<td>day-Year</td>
<td>Decades-century</td>
</tr>
<tr>
<td><strong>Stand/ecosystem model</strong></td>
<td>Emphasize how ecosystem function (physiology and growth) over time is affected by environmental climate variables given mixed species composition and vegetation structure</td>
<td>Fine .1-1 ha</td>
<td>10-100 ha</td>
<td>Hour-day</td>
<td>Years - decades</td>
</tr>
<tr>
<td><strong>Gap and Individual-tree based models</strong></td>
<td>Describe how composition and structure of vegetation changes over time in response to disturbance. Growth largely based on mortality, recruitment, and competition (light/density)</td>
<td>Fine .01-1 ha</td>
<td>10-100 ha</td>
<td>Day-decade</td>
<td>Decades-century</td>
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Table 3: A comparison of previous modeling efforts to quantify the effects of forest management on carbon sequestration.

<table>
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<td>Model used</td>
<td>Edinburgh Forest Model</td>
<td>MOSES</td>
<td>CO2FIX V.2</td>
<td>CO2FIX CCC-CCS</td>
<td>PICUS v1.4, WPM</td>
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<td>Europe/ Central America</td>
<td>Australia</td>
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<td>Stand</td>
<td>Stand</td>
<td>Stand</td>
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<td>Forest species composition</td>
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<td>Beech-Spruce Variety</td>
<td>Hemlock, Larch</td>
<td>Eucalyptus</td>
<td>Norway spruce</td>
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<td>95, 125, 50, 20</td>
<td>80</td>
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<td>90, 80</td>
</tr>
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<td>Sensitivity analysis of harvesting frequency</td>
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<td>No</td>
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<td>No</td>
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<td>Even-aged forestry</td>
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<td>Yes</td>
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<tr>
<td>Uneven-aged forestry</td>
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<td>Yes</td>
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<td>Include a “no management” scenario</td>
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<td>No</td>
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<td>Intermediate treatments</td>
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<td>2 to 5</td>
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<td>Tradeoffs of wood products</td>
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<td>Yes</td>
<td>No</td>
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1.9 Figures

Figure 1: Conceptual model of carbon reservoir and potential flux patterns within the carbon cycle. Dashed arrows represent very long temporal scales without anthropogenic influences.
Figure 2: Conceptual model showing carbon accumulation in northeastern forests throughout stand development (Oliver and Larson 1996). Note the greatest rate of carbon (C) uptake is located at the inflection point; however this is still significantly less than the maximum carbon storage potential.

Figure 3: Projected seasonal mean temperature change in the Northeast under two different emission scenarios (Frumhoff et al. 2007).
Figure 4: Model diagram of the sequential input general processing sequence of the FVS model (adapted from Dixon (2002))
Figure 5: 100 year simulation of five different regeneration scenarios projected for one stand.
CHAPTER 2: FOREST CARBON STORAGE IN THE NORTHEASTERN UNITED STATES: EFFECTS OF HARVESTING FREQUENCY AND INTENSITY INCLUDING WOOD PRODUCTS

2.1 Abstract

Temperate forests are an important carbon sink, yet there is debate regarding the net effect of forest management practices on carbon storage. Few studies have investigated the effects of different silvicultural systems, and the relative strength of *in-situ* forest carbon versus wood products pools remains in question. Our research (1) describes the impact of harvesting frequency and degree of post-harvest structural retention on carbon storage in northern hardwood-conifer forests, and (2) tests the significance of including harvested wood products in carbon accounting at the stand scale. We stratified Forest Inventory and Analysis (FIA) plots to control for environmental, forest structural, and compositional variables, resulting in 32 FIA plots distributed throughout the northeastern U.S. We used the USDA Forest Vegetation Simulator to project stand development over a 160 year period under nine different forest management scenarios. Simulated treatments represented a gradient of increasing structural retention and decreasing harvesting frequencies and included a “no harvest” scenario. The simulations incorporated carbon flux between aboveground forest biomass (dead and live pools) and harvested wood products (including carbon storage in landfills). Mean carbon storage over the simulation period, including carbon stored in harvested wood products, was calculated for each silvicultural scenario. We investigated tradeoffs among scenarios
using a factorial treatment design and two-way ANOVA. The predictive strength of management scenarios relative to site-specific variables was evaluated using Classification and Regression Trees. Mean carbon sequestration was significantly (a = 0.05) greater for “no management” compared to any of the active management scenarios. Of the harvest treatments, those favoring high levels of structural retention and decreased harvesting frequency stored the greatest amounts of carbon. In order to isolate the effect of in-situ forest carbon storage and harvested wood products, we did not include the emissions benefits associated with substituting wood fiber for other construction materials or energy sources. Modeling results from this study show that harvesting frequency and structural retention significantly affect mean carbon storage. Our results illustrate the importance of both post-harvest forest structure and harvesting frequency in carbon storage, and are valuable to land owners interested in managing forests for carbon sequestration.

2.2 Key words

Carbon, sequestration, uptake rates, additionality, wood products, structural retention, harvesting frequency, sustainable forest management, northern hardwood forests

2.3 Introduction

While deforestation accounts for 20 to 30% of total global carbon dioxide (CO₂) emissions, due primarily to tropical deforestation (IPCC 2007), forests in United States are currently a carbon (C) sink (Goodale et al. 2002), sequestering approximately 10% of
U.S. annual CO$_2$ emissions (Birdsey et al. 2006). Recognizing the important role forests play in the terrestrial C cycle and climate change mitigation efforts, developing cap and trade C markets are considering inclusion of sustainable forest management as an option for slowing rates of atmospheric CO$_2$ accumulation (Alig and Bair 2006, Canadell and Raupach 2008, Ray et al. 2009). The working hypothesis is that “improved forest management” could achieve higher levels of C storage (termed “additionality”) compared to “business as usual” or a baseline condition (Ruddell et al. 2007). While forest management clearly impacts terrestrial C storage (Birdsey et al. 2007), little information is available describing how specific forest management alternatives might affect C storage and sequestration. This understanding is vital, because the dynamics of storage and fluxes among the different sinks impacted by management (e.g. forest C pools versus wood products pools) are complex, rendering accounting of net effects on C storage challenging (Birdsey et al. 2006, Ray et al. 2009). The purpose of this study is to inform forest C management practices using empirical data coupled with forest-stand development modeling. In particular, we investigate the impact of harvested wood products in the accounting of net C sequestration in managed forests in the northeastern U.S., recognizing the pertinence of including wood products in C accounting (Seidl et al. 2007).

Some researchers have suggested that sustainably managed forests sequester more C than unmanaged forests, stressing the high tree growth rates achieved in harvested stands (Ruddell et al. 2007), and C stored in wood products (Malmsheimer et al. 2008). However, other studies have demonstrated that unmanaged forests, such as old-growth forests in the U.S. Pacific Northwest (Harmon et al. 1990, Harmon and Marks 2002) and
boreal forests in northwestern Russia (Krankina and Harmon 1994), sequester greater amounts of C than managed forests. These authors have argued that intensified forest management actually leads to a net flux of C to the atmosphere due to lower biomass in harvested stands and the often short lifespan of wood products. These conclusions, however, are based primarily on studies involving conversion of old-growth forest to young plantations (Harmon et al. 1990) and the effects of intensive harvesting practices, such as clearcutting (Krankina and Harmon 1994). Net effects on C dynamics across a range of silvicultural systems, including modified even-aged and less intensive uneven-aged forest management practices, remain poorly explored and thus are a focus of this study. We define even-aged silviculture as forest management focused on growing a single cohort of trees to financial maturity followed by harvest of merchantable timber; and uneven-aged silviculture as forest management techniques managing multiple age cohorts of trees simultaneously, with sustained harvests distributed throughout several age cohorts.

Recently, interest has developed in the use of reduced harvesting frequency (Curtis 1997) and post-harvest structural retention (Franklin et al. 1997, Keeton 2006) as approaches favoring maintenance and development of high levels of in-situ forest C storage. However, previous analyses of harvesting frequency were restricted to even-aged forest management (Liski et al. 2001, Harmon and Marks 2002, Balboa-Murias et al. 2006). None of these studies addressed the coupled effects of variations in harvesting frequency and post-harvest structural retention in mature, even to multi-aged forests, such as those now dominant on the New England landscape. Decreased harvesting frequency increases C storage in managed stands (Liski et al. 2001, Balboa-Murias et al. 2006);
however, the resulting sequestration remains less than the total C storage in unmanaged forests, even accounting for fluxes caused by natural disturbances (Krankina and Harmon 1994). The previous is restricted to certain boreal and temperate forest types. No data specifically addressing this issue are currently available for quantifying the effects of harvesting intensity on C sequestration for the northern hardwood forests of the northeastern U.S. In some studies, accounting for C stored in durable, long-lived wood products increased the estimated net C storage for intensively managed forests in which rotations periods were also increased (Perez-Garcia et al. 2005). Discrepancies among previous studies signal that further research is needed to quantify the, previously unexplored, coupled effects of harvesting frequency and intensity. This would inform the debate surrounding this issue in the forest management community (Ray et al. 2009). In this study we are particularly interested in C storage, and thus use the term “sequestration” to refer to total C stocks (forest biomass + wood products), rather than uptake rates.

Quantifying mean C sequestration under a given forest management scenario requires a temporal scale spanning at least one complete harvesting cycle. For this reason, simulation modeling is often used to quantify C sequestration in forested ecosystems. Numerous process-based, empirical, and hybrid models have been developed to project forest C dynamics in response to management activities. These studies have been conducted in a variety of forest types in Europe (Eriksson et al. 2007, Seidl et al. 2007), southeast Australia (Roxburgh et al. 2006), northwest Russia (Krankina and Harmon 1994), and northwestern (Harmon and Marks 2002) and northeastern (Neilson et al. 2006) North America. While absolute predictions generated by empirical
and hybrid models carry uncertainty, they are useful for comparing relative differences among alternate management and forest development scenarios (Zenner 2000, Eriksson et al. 2007, Seidl et al. 2007).

This study uses a widely accepted forest growth model to examine C sequestration tradeoffs among harvesting frequency and post-harvest structural retention under both even- and uneven-aged forest management, while also incorporating fluxes to wood products. We address a fundamental research question facing forest managers, namely: what is the most effective way to store C through forest management? Is C sequestration greater in more intensive approaches favoring high rates of uptake and C transfer to wood products? Or are less intensive approaches, favoring in-situ forest C storage, more effective at maximizing C storage? We test two key variables with the potential to affect forest C sequestration: 1) harvesting frequency (rotation length in even-aged silviculture and entry cycle in uneven-aged silviculture), and 2) post-harvest structural retention (residual biomass following a harvest). Our primary research objective is to inform forest C management by testing two hypotheses. The first hypothesis was that unmanaged (passive) forests would sequester greater amounts of C than actively managed forests, even accounting for C storage in durable wood products. Our second hypothesis focused on the effects of management intensity. We hypothesized that silvicultural prescriptions with increased structural retention coupled with decreased harvesting frequency would sequester the greatest amount of C relative to other active management scenarios.
2.4 Methods

2.4.1 Study area and selection of study sites

The geographic area from which forest inventory data were selected for this study is the northern hardwood region of the northeastern U.S., encompassing portions of upstate New York, Vermont, New Hampshire, and Maine (Figure 6). The study area is dominated by northern hardwood-conifer forests, in which *Acer saccharum* (sugar maple), *Fagus grandifolia* (American beech), *Tsuga canadensis* (eastern hemlock), and *Betula alleghaniensis* (yellow birch) form the major late-successional species. We used Mapmaker 2.1 (*accessed* 7/22/2008, *available at*: www.fia.fs.fed.us/tools-data/other/) to stratify the study area by eco-subregions (Bailey 2004) and then selected Forest Inventory and Analysis (FIA) plots (or sites) from within these to ensure that our sample was representative and well-distributed (Figure 6). We used the most recent FIA inventory data available at the time of this study for each state to avoid potential discrepancies among different FIA survey periods (Maine: 2003, New York: 2004, New Hampshire: 2005, Vermont: 2005). We controlled for other sources of variability by further stratifying plots based on several site-specific variables as defined in the FIA database. These included stand age (80-100 years old), slope (0 to 50%), forest type (maple-beech-birch), stand origin (natural), site productivity (site class 1-5 out of 7), physiographic class (mesic classes 21-25) basal area (BA > 23 m²/ha), and total merchantable cubic volume (> 57 m³). In order to obtain a sufficient sample size, our selection criteria encompassed a degree of heterogeneity among initial stand conditions. The stratification process resulted in a total of 32 FIA plots meeting these criteria (14 sites in the White
Mountain Region and western Maine, 3 sites in the Green Mountain Region, and 15 sites in the Adirondack Mountain Region); these were used in the model experiment and are hereafter referred to as our study sites (Table 4).

### 2.4.2 Model description

FVS was chosen for its ability to simulate forest management activities, the availability of a model variant calibrated for northern hardwoods, its availability to the general public, and its compatibility with FIA data. An additional advantage is that FVS projections are accepted by existing carbon markets. Site specific stand structure and composition data were input into FVS to project stand development under alternate management scenarios. The FVS model has been used by North American forest managers for over 30 years in a variety of applications (Bragg 2000, Wang et al. 2008), and can be used in multiple biomes (Teck et al. 1996, Crookston and Dixon 2005). FVS is effective at simulating forest growth under different management scenarios (Crookston and Dixon 2005). FVS is a distant-independent, individual tree-based forest growth model, specifically designed for and applicable to even and uneven-aged stands with simple to mixed species composition (Crookston and Dixon 2005). Aboveground biomass estimates are based on species group-specific allometric equations (Jenkins et al. 2003). The temporal scope of model projections ranges from five to several hundred years, with five-to ten-year resolution.

Component models (variants) are used to adjust models to reflect regional climatic conditions and growth rates. In this study we used the Northeast Variant (NE-FVS). NE-FVS uses growth and yield equations from NE-TWIGS (Hilt and Teck 1989),
with embedded height equations and bark ratios specific to northeastern species. No comprehensive validation study of all sub-routines within NE-FVS has been completed. However, regional validation studies of NE-FVS have shown adequate predictions of forest growth in northern hardwood forests, with model accuracy of volume predictions within 10 to 15% of actual volumes (Yaussy 2000). Modeling efficiencies of 77 to 99% were found in short term projections, however, regionally calibrated regeneration inputs are necessary to increase model accuracy in projections greater than 20 years (Bankowski et al. 1996). Furthermore, FVS is not an appropriate model for simulating impacts of climate change on forest growth (Yaussy 2000, Froese and Robinson 2007).

FVS also tracks C fluxes among wood products pools throughout product life cycles, from production to landfill or incineration, following methodologies developed by the USDA Forest Service (Smith et al. 2006). To simulate C flux in wood product pools, FVS identifies pulp and sawlogs (Dixon 2002), and applies product-specific (i.e. paper, durable wood product, etc.) life span curves based on recent data specific to North American forest types (Smith et al. 2006). In addition FVS is one of several simulation models identified by North American voluntary C markets for estimating C sequestration in managed forests as a part of climate change mitigation projects.

Our stand development simulations assumed: 1) no natural disturbances occur over time; 2) climate remains constant; and 3) C storage in soils does not change. Controlling these sources of variability helped us isolate forest management effects, and offered the opportunity to explore the relative differences between scenarios. While we recognize the uncertainty and limitations inherent to this approach, it is consistent with previous modeling work focused also on relative differences among forest management
trajectories (Harmon and Marks 2002, Eriksson et al. 2007, Seidl et al. 2007). Relative differences can remain constant even when climate change scenarios are incorporated into stand development simulations (Seidl et al. 2008).

2.4.3 Silvicultural simulations

To test our two hypotheses, we evaluated a variety of even- (Table 5) and uneven-aged (Table 6) silvicultural prescriptions. In total, we simulated nine different management scenarios in FVS, including one passive (i.e., a reserve-based) “no management” scenario and eight active management scenarios. The latter were representative of silvicultural systems used commonly in the Northeast, but were modified to encompass a range of harvesting intensities. Specific parameters of prescriptions were derived from experience and studies in the Northeast (Leak et al. 1986, Seymour 1995, Nyland 1996, 1998, Keeton 2006). Silvicultural prescriptions used in this study included four even-age scenarios and four uneven-age scenarios. Within these broad silvicultural groups, individual treatments were derived by factoring two levels for each of two categories: harvesting frequency and degree of structural retention (Table 5 and Table 6), for a total of 8 active management scenarios.

To test the effect of harvesting frequency on C sequestration, stand development simulations for the four active management scenarios were run under two different harvesting intervals, one long (120 years for even-aged scenarios; 30 years for uneven-aged scenarios) and one short (80 years for even-aged scenarios; 15 years for uneven-aged scenarios) (Tables 5 and 6).

To evaluate the effect of structural retention, we developed two different even-
aged management scenarios representing different levels of structural retention. A clearcut represented low structural retention, with a complete removal of all trees greater than 5 cm diameter at breast height (DBH), and all harvesting residue (slash) removed from the site (Table 5). A shelterwood (Nyland 1996) represented high structural retention, with the retention of six legacy trees (canopy trees never harvested) per hectare and all slash left on site (Table 5). In uneven-aged scenarios, two individual tree selection (ITS) systems were used. In ITS systems, harvesting was based on a pre-defined diameter distribution (q factor) that directed harvesting towards diameter classes with stem densities above target levels (Table 6). The first ITS represented low retention, where at each entry the stand was harvested to a residual basal area of 15 m$^2$/ha, with no retention of large trees over the maximum diameter of 50 cm diameter used to define the target diameter distribution. The second ITS represented high retention, where at each entry the stand was harvested to a residual basal area of 19 m$^2$/ha, with the retention of 12 large trees over the maximum diameter of 61 cm used to define the target diameter distribution.

2.4.4 Regeneration inputs in model simulations

We ran all the management scenarios over 160 year simulation periods in order to capture a minimum of one complete harvesting cycle. Model calculations (e.g., predicted growth and mortality) were performed on five year time steps (Dixon 2002). Because NE-FVS includes only a vegetative regeneration sub-model (i.e., limited stump sprouting only), user-defined regeneration parameters (including species, spatial distribution, total number per acre, and seedling size) must be defined in order to simulate sexual
reproduction. Natural regeneration rates in northern hardwood forests were acquired from the literature (Graber and Leak 1992), and field data for similar silvicultural treatments and site/stand conditions (Keeton unpublished data) (Table 7). These natural regeneration rates were used to develop background regeneration rates based on average site species composition. Background regeneration rates were used to emulate natural regeneration within stands, independent of forest management activities.

For active management scenarios, we adapted regeneration data specific to northern hardwood even-aged forest management (Leak 1987, 2005) and uneven-aged forest management (Mader and Nyland 1984, Leak 1987). Input regeneration values (Table 7) were correlated with percent canopy cover (i.e., decreased percent canopy cover as a result of harvesting activities increases total number of seedlings per hectare). We also adjusted the relative proportions of intermediate vs. tolerant species based on percent canopy cover. Management scenario-specific regeneration values were input at the time step immediately following all simulated regeneration harvests. Model sensitivity of aboveground biomass accumulation to regeneration input was tested using a series of five regeneration input ranges, including one simulation with no regeneration, one based on literature values, and three with adjusted literature values. This sensitivity analysis was performed for each management scenarios. In NE-FVS, substantially increased regeneration results in an early leveling off of biomass accumulation. This is because mortality is modeled exclusively as a function of total stand density. Consequently, we reduced literature derived values proportionally to percent canopy removal, in order to reflect realistic biomass accumulation over a 160 year simulation period.
2.4.4 Data analysis

Simulation output from the 32 different sites were averaged to produce mean values for each scenario. All values, unless stated otherwise, are presented as mean C sequestration over the 160 year simulation period. We calculated the mean C stock in aboveground biomass (live and dead) and wood products during the simulation period, as a way to compare C sequestration between different management scenarios (Eriksson et al. 2007). In order to test our first hypothesis, examining the tradeoffs in C sequestration between active and passive management, we used SPSS 16.0 (2008) statistical software to run single-factor ANOVA and post-hoc Bonferroni multiple comparisons to test for significant differences (a = 0.05) between scenarios. To address our second hypothesis, examining the effect of management intensity on C sequestration, we used two-way ANOVA to test for the significance of harvesting frequency, structural retention, and the interaction between the two relative to mean C sequestration.

We also conducted a sensitivity analysis to help identify subtle differences in the effects of harvesting frequency on C sequestration. We did this by adjusting the low and high harvesting frequency scenarios applied to each of the four original silvicultural prescriptions. The original high harvesting frequency (80 years in even-aged and 15 years in uneven-aged scenarios) was decreased by 25% to create two additional harvesting frequencies (60 years for even-aged and 11 years for uneven-aged). The original low harvesting frequency (120 years in even-aged and 30 years in uneven-aged) was increased by 25% to create two additional harvesting frequencies (150 years for even-aged and 38 years for uneven-aged scenarios). Due to processing limitations in the
model, we were unable to simulate extremely high harvest frequencies (harvesting frequency < 15) for uneven-aged scenarios over the entire 160 year simulation period. For this reason, the 25% below original high frequency (11 year entry cycles) for uneven-aged management are computed in FVS the same as the original high frequency (15 year harvesting frequency), and the sensitivity analysis in uneven-aged scenarios is restricted to three different harvesting frequencies (15, 30, and 38 years). Adjusted model outputs were tested using two-way ANOVA.

A logical criticism of attributing predicted C sequestration effects solely to management scenario is that certain site characteristics, such as productivity, pre-harvest stand volume, and species composition (e.g., percent conifer), might also affect forest growth rates and C sequestration potential. To evaluate this, we used a classification and regression tree (CART) to test the predictive strength of management scenarios relative to other site-specific environmental, structural, and compositional characteristics, modeled as independent variables. CART analysis is recognized as a powerful tool for analyzing complex ecological data (De'ath and Fabricius 2000). CART is a robust, nonparametric, binary method that partitions variance in a response variable through a series of repeated splits (branches) based on the values of independent variables (Breiman et al. 1984, Keeton et al. 2007: p. 857). CART was chosen for its ability to explain the variation of a single response variable (in this case, mean C sequestration) based on multiple categorical or continuous independent variables (De'ath and Fabricius 2000). We used both categorical and continuous independent variables from original FIA plot measurements (Table 8). To avoid redundancy among predictor variables we tested all independent variables for collinearity. Independent variables exhibiting strong
collinearity ($r^2 > 0.60$) were not included in further analyses. CART analysis was performed using S-Plus software (Statistical Sciences 2002). Cost-complexity pruning was used to eliminate non-significant nodes. Pruning was dictated by $\alpha = 0.05$, a measure of how much additional accuracy an individual split must add to the entire tree to warrant the additional complexity. This process controls the tree size, and limits tree complexity.

### 2.5 Results

#### 2.5.1 Mean C sequestration under alternate forest management scenarios

*Simulation model predictions*

The simulation results show a clear gradient of C sequestration ranging from high intensity forest management (clearcut) to low intensity management (ITS_HighLow and No Management) (Figure 7). Sharp declines in C within active management scenarios are caused by the removal of C from the forest following a scheduled harvest. The amplitude of these declines is muted by the flux of C into storage pools in wood products, as well as, the averaged 10-year C sequestration values. Generally, scenarios with decreased harvesting frequency show greater accrual of C as a result of accretion of C in dead wood pools and increased live biomass (Figure 7). Clearcut scenarios sequestered less C than all other management scenarios (Table 9). Shelterwood scenarios sequestered similar amounts of C as ITS scenarios emphasizing low structural retention. Of the active management scenarios, ITS scenarios incorporating high structural retention sequestered the greatest amount of C (Table 9). Mean C sequestration in the no
management scenario was significantly higher ($p < 0.01$) than all other scenarios shown by the ANOVA and Bonferroni post-hoc test (Figure 8).

**Effects of harvesting frequency and intensity**

Model predictions showed that harvesting intensity significantly affected C sequestration ($p < 0.01$), based on the results of the two-way ANOVA. In our initial analysis, harvesting frequency did not have a statistically significant effect ($p = 0.081$, Table 10). The interactive effect of harvesting frequency and retention also was not statistically significant ($p = 0.584$). In order to investigate more subtle differences among silvicultural prescriptions, we re-ran the two-way ANOVAs, separating treatments into two groups: even-aged (clearcut and shelterwood scenarios) and uneven-aged treatments (ITS scenarios) (Table 10). The second iteration of the two-way ANOVA showed that in uneven-aged scenarios harvesting frequency significantly affected C sequestration ($p = 0.01$). Conversely, in even-aged scenarios, given our initial harvesting frequencies (80 and 120 year harvesting cycles), harvesting frequency did not significantly affect C sequestration ($p = 0.658$). In both uneven and even-aged scenarios, retention significantly affected C sequestration ($p < 0.01$). Furthermore, the interaction of harvesting frequency and retention was not significant in either uneven-aged ($p = 0.716$) or even-aged ($p = 0.554$) management scenarios.

To test model sensitivity to harvesting frequency, we performed a secondary analysis in which we adjusted harvesting frequency in all active management scenarios. A third two-way ANOVA analysis was done to test the effects of the adjusted harvesting frequencies on mean C sequestration within management scenarios (Table 11).
Harvesting frequency significantly (a = 0.05) affected C sequestration in all adjusted scenarios (p = 0.01) in which the time difference between low and high frequencies was increased by 25% or more. In all scenarios the interaction of harvesting frequency and structural retention was not significant (p > 0.01), except when scenarios were compared against even-aged scenarios with harvesting frequency set to 60 years (p < 0.01). In this case, the strong interaction was driven by a combination of extremely high harvesting frequencies (relative to typical silvicultural practices in the northern hardwood region), and very low structural retention.

**Effects of forest management scenario versus site-specific factors**

The CART results (N = 288) strongly supported our second hypothesis that harvesting frequency and intensity significantly affect C sequestration, but showed that site specific variables, in some cases, can also be important secondary predictors. Of the eleven independent variables included in the initial model, four variables were incorporated in the final CART model: management scenario, site index, percent conifer, and basal area. Of these variables, management scenario was the strongest predictor of mean C sequestration in CART models, explaining variance at both primary, and in some cases, lower splits on the tree (Figure 9). The primary split at the root node, or top of the tree, is divided between active and passive management techniques (Figure 9). The left side of the tree is further divided at the next node between high intensity (higher harvesting frequency and lower retention) and low intensity (lower harvesting frequency and higher retention) active management scenarios. However, after the general range of C sequestration potential is established by management scenario, CART showed that
some sub-groupings of sites with higher site index (i.e. more productive), greater initial basal area (e.g. > 36.4 m²/ha), and lower percent conifer (e.g. < 15%) will have significantly greater mean C sequestration. Together these results indicate the potential for interaction between management scenario and site specific conditions.

2.5.2 Effects of forest management scenarios on C uptake rates

To clarify the relative importance of uptake rates versus storage in our estimates of total predicted sequestration, we calculated annual C uptake rates three different ways (Table 12): 1) C uptake rate per harvest cycle with the inclusion of wood products; 2) C uptake rate per simulation without the inclusion of C stored in wood products; and 3) C uptake rate per simulation with the inclusion of wood products. When C uptake rates were averaged by management scenario, clearcut scenarios had greater C uptake rates than all other scenarios (clearcut uptake rate: high harvesting frequency = 0.55 Mg C·ha⁻¹·yr⁻¹, and low harvesting frequency = 0.44 Mg C·ha⁻¹·yr⁻¹). C uptake rates in the no management scenario were the third highest overall (uptake rate = 0.36 Mg C·ha⁻¹·yr⁻¹). When averaged over the 160 year simulation period without the inclusion of C stored in wood products, C uptake rates in three scenarios were negative (shelterwood_low = -0.02 Mg C·ha⁻¹·yr⁻¹, ITS_LowHigh = -0.02 Mg C·ha⁻¹·yr⁻¹, ITS_LowLow = -0.04 Mg C·ha⁻¹·yr⁻¹). However, the inclusion of C stored in wood products resulted in positive uptake rates for all scenarios. It should be noted that mean C uptake rates for the 160 year simulation period include harvesting activities, wherein significant amounts of C are lost from forest pools following treatment.
2.6 Discussion

Forest management intensity strongly affects C sequestration based on our results. While our findings tell a novel story, they build on previous studies conducted throughout the world’s temperate forested regions (Roxburgh et al. 2006, Schmid et al. 2006, Eriksson et al. 2007, Seidl et al. 2007). Previous research in Australia showed that actively managed forests can sequester substantial amounts of C and should be considered when developing terrestrial C management options (Roxburgh et al. 2006). Furthermore, research in European temperate forests has shown the importance of considering wood products in C accounting (Schmid et al. 2006, Eriksson et al. 2007, Seidl et al. 2007). Unlike previous studies, our results show there can be important interactive effects of post-harvest structural retention and harvesting frequency. These findings are relevant to ongoing debates regarding forest management and C sequestration, as addressed by our two hypotheses. The results supported both our first hypothesis that passive management sequesters more C than active management, as well as, our second hypothesis that management practices favoring lower harvesting frequencies and higher structural retention sequester more C than intensive forest management.

Currently, the incorporation of active forest management in climate change mitigation is widely debated. On one hand, intensively managed forests with high harvesting frequencies that produce wood products and biofuels are recognized as a viable option for reducing C emissions through the substitution of more C intensive products or energy (Eriksson et al. 2007, Malmsheimer et al. 2008). On the other hand,
numerous studies have concluded that the replacement of older forests with younger forests results in a net increase in C released to the atmosphere (Cooper 1983, Harmon et al. 1990, Schulze et al. 2000). Our results support these latter findings, and show that a shift towards intensively managed forests does not increase C sequestration when C accounting is restricted to C sequestration in aboveground forest biomass and harvested wood products.

2.6.1 Effects of forest management on carbon sequestration

Our study is among the first to explore the combination of both harvesting intensity and frequency. Model predictions showed that management practices favoring lower harvesting frequencies and higher structural retention sequester more C than more intensive forest management practices. In addition, we conclude there are more nuanced effects of structural retention and harvesting frequency based on the results. In our first iteration of management scenario projections, structural retention had a greater effect on C sequestration than harvesting frequency. However, our sensitivity analysis showed that harvesting frequency can significantly affect C sequestration when rotation periods are sufficiently extended (or differentiated in the case of our methodology). This finding is supported by prior research (Krankina and Harmon 1994, Liski et al. 2001, Balboa-Murias et al. 2006). Unlike previous studies focused on even-aged management (Harmon et al. 1990, Liski et al. 2001, Balboa-Murias et al. 2006) or in-situ forest carbon without consideration of wood products (Krankina and Harmon 1994), our analysis demonstrated the importance of retention and harvesting frequency for both even- and uneven-aged silvicultural practices with the inclusion of wood products. Furthermore, we could expect
the differences between intensive and less intensive management to be even greater with
the inclusion of energy inputs (i.e., diesel fuel and gasoline) associated with timber
harvesting, trucking, and processing.

Accounting for emissions offsets from the substitution of wood products for non-
wood products, such as steel and concrete, can significantly change the net C effect of
forest management (Hennigar et al. 2008). This is especially true when considering the
potential for reduced availability of wood products associated with decreased harvesting
(Ray et al. 2009). Comprehensive life-cycle analyses show that substituting wood
products for steel and concrete decreases emissions of CO\textsubscript{2} to the atmosphere, due to the
energy inputs required to manufacture the latter (Lippke et al. 2004). However,
incorporation of substitutive effects within life-cycle analyses is challenging and
potentially unreliable due to uncertainties in quantifying emissions from wood products
transportation and methane emissions attributable to decomposition of forest products in
landfills (Miner and Perez-Garcia 2007). Studies focusing on the substitutive benefits
associated with wood products suggest that if the sole goal of forest management is to
sequester C (and not to restrict C storage to forest C pools), both high frequency intensive
management and low frequency less intensive management can be equivalent under
certain conditions (Malmsheimer et al. 2008). However, these conclusions are not based
on analysis across a spectrum of forest management scenarios, for instance encompassing
both uneven-aged and even-aged silviculture. Moreover, C markets currently only award
credits for C stored in the forest and in wood products due to the complexities involved
with broader energy accounting (Ruddell et al. 2007). It is critical to understand the
individual impacts of fluxes between pools in order to inform broader studies addressing
Relative to even-aged management, few studies have investigated the effects of harvesting frequency on C sequestration in uneven-aged silviculture. Our study showed that for uneven-aged management scenarios common to the northern hardwood region of eastern North America, decreased harvesting frequency significantly increased C sequestration, independent of post-harvest structural retention. These findings suggest that decreasing harvesting frequency alone may not be effective for enhancing forest C storage in this region. In addition, there was a significant interaction between very high harvesting frequency and post-harvest structural retention for even-aged forestry. Thus, consideration of both structural retention and harvesting frequency is necessary to optimize forest C sequestration in northern hardwood ecosystems.

2.6.2 Carbon uptake rates versus storage

Another important issue is the relative importance of C uptake rates versus in-situ storage (or biomass) in terms of effects of total ecosystem sequestration (Fahey et al. 2005). Our results showed that increased management intensity was positively correlated with increased C uptake rates. Younger forests have high C uptake rates, though they store significantly less C than older forests (Harmon et al. 1990, Harmon 2001, Luyssaert et al. 2008). However, C uptake rates vary depending on the scale (spatial, temporal, and process resolution) at which they are measured or assessed (Harmon 2001). Our results showed that when the temporal scope was restricted to one harvesting cycle, the greatest C uptake rates were in clearcut scenarios (0.55 Mg C·ha\(^{-1}\)·yr\(^{-1}\) and 0.44 Mg C·ha\(^{-1}\)·yr\(^{-1}\)), representing the highest intensity management scenario. These findings are consistent
With previous research (Hoover and Stout 2007).

With the exception of the two clearcut scenarios, the “no management” scenario had greater C uptake rates than all other management scenarios. We believe this is a result of two factors: 1) model sensitivity to regeneration inputs; 2) a net increase in C sequestered in dead wood pools. We examined the first factor by testing model sensitivity to varying regeneration inputs, confirming the model’s high sensitivity to user-defined regeneration inputs. Model sensitivity to regeneration was tested by re-running all 32 stands in two randomly selected management scenarios with no regeneration inputs. Results from these two additional simulations showed large increases in C uptake rates (up to 12.5 times greater). Mortality and stand developmental dynamics within FVS are largely a function of stand density; hence, accurate regeneration inputs are critical. NE-FVS simulations lacking well researched, user-defined regeneration inputs do not realistically reflect stand developmental processes for northern hardwood forests.

To address the influence of dead wood accumulation on uptake rates, we analyzed model partitioning of C within forest pools (Table 9). In the “no management” scenario dead wood recruited and accumulated for longer and at faster rates compared to management scenarios, with C additions to dead wood pools exceeding C losses from decomposition. Allocation of C to dead wood pools increases with forest stand development and, in some cases, compensates for declining growth rates in older trees in terms of total ecosystem biomass accumulations (Harmon 2001, Franklin et al. 2002, Goodale et al. 2002). For this reason, in our results “no management” had C accrual rates similar to the highest C accrual rates seen in intensive active management scenarios, where rapid biomass accretion was closely related to increased growth rates. Excepting
the most intensive management scenarios (i.e., clearcutting), our results did not show that higher frequency, intensively managed forests have greater total C accumulation rates than older, slower growing forests. We attribute this to a combination of model sensitivity to regeneration, projected net positive C additions in live trees (Hadley and Schedlbauer 2002, Keeton et al. 2007, Luyssaert et al. 2008), and the significantly greater dead wood C pool that develops over time under less intensive management scenarios. Harmon (2001) suggested that the specific parameters used to assess C sequestration dynamics can profoundly influence scenario comparisons, an assertion supported by our model results.

2.6.3 Uncertainty in projections

We recognize the uncertainties within model predictions that result from underlying assumptions. Fine-scaled canopy disturbance is the dominant disturbance type in the Northeast (Seymour et al. 2002), and occur on return intervals of 50 to 200 years (Runkle 1982). Disturbance regimes impact C sequestration through rapid flux of C from living biomass to dead wood pools following large-scale disturbance (McNulty 2002), or more gradual flux of C between pools as a result of small to intermediate-scale disturbances (Thurig et al. 2005). Furthermore, climate change is likely to cause individual species range shifts (Beckage et al. 2008), community compositional changes (Xu et al. 2009), and increased mortality from drought and disease (van Mantgem et al. 2009). Other research has focused on the incorporation of climate change into model projections of forest ecosystem processes (Aber et al. 2001), however, this was not within the scope of this project. Changes in climate and natural disturbance regimes will
inevitably impact forests of the Northeast in the next 160 years. Understanding the relative differences of forest management practices independent of these processes is an important first step in understanding the effects of forest management practices on C sequestration; however, the potential impacts of climate change and natural disturbance should not be overlooked, and adaptive management practices that respond to these impacts are recommended.

2.6.4 Integrating carbon sequestration into forest management systems

There is significant potential for enhanced carbon sequestration by modifying harvesting frequencies and retention levels, applied both to conventional silvicultural systems (Nyland 1996) as well as innovative systems, such as disturbance-based forestry (North and Keeton 2008). Some silvicultural tools have already been developed that utilize these concepts and would be applicable for land managers interested in managing for increased C sequestration. In the U.S. Pacific Northwest, for example, the variable retention harvest system (Franklin et al. 1997) retains post-harvest biomass and better approximates natural disturbance effects, including persistence of biological legacies (Franklin et al. 2002). In the U.S. Northeast, silvicultural approaches that emulate the frequency and scale of natural disturbances (Seymour et al. 2002, Seymour 2005), and increase post-harvest structural retention (Keeton 2006) represent options for managing for high biomass forests. In temperate European forests, conversion from short rotation, even-aged forestry to uneven-aged management has been shown to increase net C sequestration, even under multiple climate change scenarios (Seidl et al. 2008). Less intensive management strategies may provide co-varying ecosystem services, such as
enhanced habitat for late successional wildlife biodiversity (McKenny et al. 2006, Smith et al. 2008), hydrologic regulation (Jackson et al. 2005), and riparian functionality (Keeton et al. 2007).

2.6.5 Conclusion: implications for carbon market participation

Sustainably managed forests sequester considerable amounts of C and thus have a role to play in climate change mitigation projects (Ruddell et al. 2007). However, it is essential to recognize that forestry is only one of many necessary abatement options (Tavoni et al. 2007). Standardized protocols for both managing and measuring C in forests are necessary to achieve demonstrable C sequestration benefits (Lindner and Karjalainen 2007), while maintaining socially (Agrawal et al. 2008) and ecologically (Chazdon 2008) responsible mitigation projects. The methodologies used in this study provide a simple framework, with broad geographic applicability, for assessing C sequestration effectiveness in managed forests. With nationally available FIA data, and a widely accessible simulation model, our general methodology can be replicated in other regions. Findings from this study together with further research will help policy makers evaluate the potential for forest management to contribute to climate mitigation programs.

Emerging cap and trade C markets may provide a potential source of revenue for forest owners interested in practicing sustainable forest management (Ray et al. 2009). To benefit from this opportunity, landowners will have to demonstrate a change in management leading to enhanced C sequestration or “additionality.” Our findings suggest that passive or less intensive management are the most effective management
techniques for achieving additionality, assuming no inclusion of substitution effects. We showed that even with consideration of C sequestered in harvested wood products, unmanaged northern hardwood forests will sequester a minimum of 28% more C than any of the active management options evaluated. This finding suggests that reserve-based approaches will have significant carbon storage value.

However, this does not mean that additionality cannot also be achieved through specific choice of active forest management approach. For example, we showed that a shift in management from high intensity, high frequency management to low intensity, low frequency management can sequester up to 26% more C. This difference is largely a result of the significant initial loss of C incurred from removal of large quantities of C stored in live and dead tree biomass, slow post-harvest accretion of C in dead wood pools, and the transient nature of C in the wood product stream (Smith et al. 2006). Collectively, our findings suggest that a shift to less intensive forest management alternatives will result in a net increase in C sequestration in northern hardwood ecosystems, so long as the accounting is restricted to forest and wood products C pools.

2.7 Acknowledgments

This research was supported by grants from the Northeastern States Research Cooperative, and the USDA McIntire-Stennis Forest Research Program. The authors are grateful to graduate students in Geology 371 at the University of Vermont who provided critical feedback that greatly improved this manuscript. Helpful reviews were also provided by Jennifer Jenkins and Shelly Rayback at the University of Vermont.
### 2.8 Tables

Table 4: Environmental, structural, and compositional characteristics of the 32 Forest Inventory and Analysis (FIA) plots used in simulation modeling.

<p>| FIA Plot Code | Starting Stand Age | Ecosystem ** | Site Index | Slope (%) | Elevation (meters) | Aspect (degrees) | Percent Conifer (% BA) | Basal Area (m²/ha) | SDI | Trees per Hectare | QMD (m³·ha⁻¹·yr⁻¹) | MAI (m²·ha⁻¹·yr⁻¹) | Number of Strata * | Canopy Height (m) | Percent Canopy Cover |
|---------------|---------------------|--------------|------------|-----------|-------------------|------------------|------------------------|-------------------|-----|-----------------|----------------|----------------||-----------------|-----------------|---------------------|
| 2320030702501505 | 94 | M211Af | 44 | 14 | 518 | 195 | 13 | 37.6 | 510 | 10843 | 2.6 | 2.6 | 1 | 18.6 | 80 |
| 2320030702502686 | 97 | M211Af | 42 | 12 | 427 | 235 | 21 | 31.5 | 444 | 11125 | 2.4 | 1.6 | 1 | 19.5 | 82 |
| 2320030900702261 | 86 | M211Af | 34 | 8 | 549 | 215 | 34 | 33.1 | 506 | 17432 | 1.9 | 1.8 | 1 | 19.2 | 76 |
| 2320030900703046 | 80 | M211Ae | 42 | 9 | 701 | 100 | 18 | 30.5 | 480 | 18318 | 1.8 | 2.2 | 1 | 17.4 | 73 |
| 2320030900703313 | 87 | M211Ag | 51 | 12 | 183 | 2 | 50 | 35.1 | 430 | 5997 | 3.4 | 2.5 | 1 | 17.1 | 80 |
| 2320030900703677 | 89 | M211Af | 81 | 10 | 488 | 140 | 1 | 26.2 | 384 | 11191 | 2.1 | 1.6 | 1 | 19.5 | 79 |
| 2320030901700110 | 84 | M211Ag | 37 | 14 | 366 | 22 | 62 | 42.2 | 604 | 16032 | 2.3 | 3.2 | 2 | 21.3 | 72 |
| 2320030901700852 | 81 | M211Af | 37 | 13 | 823 | 248 | 42 | 29.4 | 372 | 6005 | 3.1 | 1.9 | 1 | 16.2 | 59 |
| 2320030901701013 | 96 | M211Ae | 41 | 14 | 610 | 124 | 17 | 34.7 | 450 | 8058 | 2.9 | 2.4 | 1 | 18.6 | 69 |
| 2320030901702963 | 85 | M211Ag | 65 | 27 | 274 | 65 | 0 | 24.6 | 334 | 7117 | 2.6 | 1.8 | 2 | 21.3 | 78 |
| 3320050200300163 | 82 | M211Ad | 81 | 17 | 274 | 250 | 0 | 30.5 | 398 | 7122 | 2.9 | 2.9 | 1 | 24.4 | 78 |
| 3320050200700781 | 80 | M211Af | 62 | 5 | 549 | 60 | 22 | 28.7 | 355 | 5300 | 3.3 | 2.3 | 1 | 21.9 | 71 |
| 3320050200900018 | 85 | M211Ba | 83 | 12 | 579 | 343 | 0 | 26.6 | 395 | 11826 | 2.1 | 2.8 | 1 | 26.8 | 73 |
| 3320050200900904 | 97 | M211Ad | 49 | 3 | 427 | 0 | 34 | 32.6 | 454 | 10939 | 2.4 | 2.1 | 1 | 23.5 | 82 |
| 3620040303506767 | 81 | M211Db | 62 | 0 | 335 | 0 | 44 | 47.8 | 477 | 2894 | 5.7 | 4.6 | 1 | 23.2 | 86 |
| 3620040304303762 | 80 | M211Dd | 60 | 12 | 457 | 179 | 3 | 38.1 | 465 | 6440 | 3.4 | 3.5 | 1 | 24.4 | 82 |
| 3620040304303966 | 80 | M211Dd | 43 | 6 | 549 | 256 | 27 | 33.1 | 403 | 5545 | 3.4 | 2.4 | 1 | 21.3 | 85 |
| 3620040304310188 | 95 | M211Df | 46 | 16 | 640 | 85 | 18 | 29.8 | 437 | 12639 | 2.2 | 2.1 | 1 | 24.4 | 71 |
| 3620040304312007 | 92 | M211Df | 88 | 20 | 549 | 81 | 4 | 30.5 | 354 | 4040 | 3.9 | 2.5 | 1 | 25.9 | 76 |
| 3620040304312851 | 97 | M211Df | 35 | 18 | 335 | 148 | 37 | 35.1 | 413 | 4982 | 3.7 | 2.4 | 1 | 20.1 | 79 |
| 36200403043105127 | 100 | M211Df | 50 | 13 | 701 | 287 | 7 | 24.6 | 330 | 6808 | 2.7 | 1.5 | 1 | 20.1 | 66 |
| 36200403043105218 | 90 | M211Df | 57 | 33 | 305 | 137 | 57 | 33.5 | 443 | 8599 | 2.8 | 2.1 | 1 | 21.0 | 75 |
| 3620040404102413 | 82 | M211Dd | 47 | 0 | 640 | 0 | 15 | 48.0 | 525 | 4663 | 4.5 | 4.8 | 1 | 25.3 | 75 |
| 3620040404102456 | 86 | M211Dd | 60 | 12 | 671 | 12 | 15 | 29.6 | 362 | 5115 | 3.4 | 2.3 | 1 | 25.0 | 73 |</p>
<table>
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<th>Site</th>
<th>Code</th>
<th>Species</th>
<th>Age</th>
<th>Stage</th>
<th>DBH</th>
<th>Height</th>
<th>DBH Error</th>
<th>Height Error</th>
<th>DBH Std Err</th>
<th>Height Std Err</th>
<th>Age Std Err</th>
<th>Stage Std Err</th>
<th>Notes</th>
</tr>
</thead>
<tbody>
<tr>
<td>3620040404102703</td>
<td>90</td>
<td>M211Dd</td>
<td>62</td>
<td>18</td>
<td>579</td>
<td>327</td>
<td>57</td>
<td>26.2</td>
<td>345</td>
<td>6588</td>
<td>2.8</td>
<td>2.0</td>
<td></td>
</tr>
<tr>
<td>3620040404104669</td>
<td>91</td>
<td>M211Dd</td>
<td>41</td>
<td>22</td>
<td>732</td>
<td>306</td>
<td>20</td>
<td>29.2</td>
<td>363</td>
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<td>3.2</td>
<td>2.1</td>
<td></td>
</tr>
<tr>
<td>3620040404106138</td>
<td>86</td>
<td>M211Dd</td>
<td>60</td>
<td>12</td>
<td>579</td>
<td>12</td>
<td>27</td>
<td>38.3</td>
<td>480</td>
<td>7480</td>
<td>3.2</td>
<td>3.2</td>
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</tr>
<tr>
<td>3620040411302486</td>
<td>80</td>
<td>M211De</td>
<td>88</td>
<td>12</td>
<td>488</td>
<td>166</td>
<td>0</td>
<td>44.3</td>
<td>506</td>
<td>5382</td>
<td>4.0</td>
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<tr>
<td>3620040411305029</td>
<td>100</td>
<td>M211De</td>
<td>48</td>
<td>14</td>
<td>518</td>
<td>169</td>
<td>51</td>
<td>25.5</td>
<td>357</td>
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<td>1.8</td>
<td></td>
</tr>
<tr>
<td>5020050200900479</td>
<td>91</td>
<td>M211Ae</td>
<td>37</td>
<td>11</td>
<td>396</td>
<td>276</td>
<td>44</td>
<td>38.8</td>
<td>507</td>
<td>9160</td>
<td>2.9</td>
<td>3.0</td>
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<tr>
<td>5020050201701120</td>
<td>85</td>
<td>M211Ba</td>
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<td>235</td>
<td>0</td>
<td>29.6</td>
<td>400</td>
<td>828</td>
<td>2.7</td>
<td>2.4</td>
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</tr>
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<td>81</td>
<td>M211Ca</td>
<td>89</td>
<td>47</td>
<td>183</td>
<td>10</td>
<td>0</td>
<td>23.0</td>
<td>261</td>
<td>2743</td>
<td>4.1</td>
<td>2.9</td>
<td></td>
</tr>
</tbody>
</table>

Note: All values were measured by USDA Forest Service Forest Inventory and Analysis Program, and retrieved through the stand list file in FVS.
* As defined in Crookston and Stage 1999
** As defined in Cleland et al. 1997
Table 5: Description of the four even-aged silvicultural prescriptions used as management scenarios. We used a factorial design to test the independent effects of and interactions among two levels each for harvesting frequency and structural retention.

<table>
<thead>
<tr>
<th>Harvesting Frequency</th>
<th>High (80 years)</th>
<th>Low (120 years)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Clearcut_High</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1) Commercial thin: implement when stand reaches stocking density above fully stocked</td>
<td>1) Commercial thin: implement when stand reaches stocking density above fully stocked</td>
<td></td>
</tr>
<tr>
<td>Number of permanently retained trees/ha: 0</td>
<td>Number of permanently retained trees/ha: 0</td>
<td></td>
</tr>
<tr>
<td>Slash removed from site</td>
<td>Slash removed from site</td>
<td></td>
</tr>
<tr>
<td><strong>Clearcut_Low</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Shelterwood_High</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1) Commercial thin: implement when stand reaches stocking density above fully stocked</td>
<td>1) Commercial thin: implement when stand reaches stocking density above fully stocked</td>
<td></td>
</tr>
<tr>
<td>2) Shelterwood: 2005 and 2085</td>
<td>2) Shelterwood: 2005 and 2125</td>
<td></td>
</tr>
<tr>
<td>Residual basal area: 14 m²/ha</td>
<td>Residual basal area: 14 m²/ha</td>
<td></td>
</tr>
<tr>
<td>Number of permanently retained trees/ha: 6</td>
<td>Number of permanently retained trees/ha: 6</td>
<td></td>
</tr>
<tr>
<td>Smallest diameter in removal cut: 15 cm</td>
<td>Smallest diameter in removal cut: 15 cm</td>
<td></td>
</tr>
<tr>
<td>Slash left on site</td>
<td>Slash left on site</td>
<td></td>
</tr>
<tr>
<td><strong>Shelterwood_Low</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Residual Structure</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Low</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Table 6: Description of the four different uneven-aged silvicultural prescriptions used as management scenarios. We used a factorial design to test the independent effects of and interactions among two levels each for harvesting frequency and structural retention.

<table>
<thead>
<tr>
<th>Residual Structure</th>
<th>High (15 years)</th>
<th>Low (30 years)</th>
</tr>
</thead>
<tbody>
<tr>
<td>ITS_LowHigh</td>
<td>Q-value*: 1.3</td>
<td>Q-value*: 1.3</td>
</tr>
<tr>
<td>Residual basal area: 15 m²/ha</td>
<td>Residual basal area: 15 m²/ha</td>
<td></td>
</tr>
<tr>
<td>Min DBH class: 5 cm</td>
<td>Min DBH class: 5 cm</td>
<td></td>
</tr>
<tr>
<td>Max DBH class: 50 cm</td>
<td>Max DBH class: 50 cm</td>
<td></td>
</tr>
<tr>
<td>DBH class width: 5 cm</td>
<td>DBH class width: 5 cm</td>
<td></td>
</tr>
<tr>
<td>Number of legacy trees/ha†: 0</td>
<td>Number of legacy trees/ha†: 0</td>
<td></td>
</tr>
<tr>
<td>Slash left on site</td>
<td>Slash left on site</td>
<td></td>
</tr>
<tr>
<td>ITS_HighHigh</td>
<td>Q-value*: 1.3</td>
<td>Q-value*: 1.3</td>
</tr>
<tr>
<td>Residual basal area: 19 m²/ha</td>
<td>Residual basal area: 19 m²/ha</td>
<td></td>
</tr>
<tr>
<td>Min DBH class: 5 cm</td>
<td>Min DBH class: 5 cm</td>
<td></td>
</tr>
<tr>
<td>Max DBH class: 61 cm</td>
<td>Max DBH class: 61 cm</td>
<td></td>
</tr>
<tr>
<td>DBH class width: 5 cm</td>
<td>DBH class width: 5 cm</td>
<td></td>
</tr>
<tr>
<td>Number of legacy trees/ha†: 12</td>
<td>Number of legacy trees/ha†: 12†</td>
<td></td>
</tr>
<tr>
<td>Average diameter of legacy tree: 41 cm</td>
<td>Average diameter of legacy tree: 41 cm</td>
<td></td>
</tr>
<tr>
<td>Slash left on site</td>
<td>Slash left on site</td>
<td></td>
</tr>
<tr>
<td>ITS_HighLow</td>
<td>Q-value*: 1.3</td>
<td>Q-value*: 1.3</td>
</tr>
</tbody>
</table>

* Q-value is defined as the ratio of the number of stems to those in each successively larger diameter class

† Legacy tree is defined as a permanently retained tree larger than the maximum diameter used to define the target diameter distribution
Table 7: Regeneration inputs used in model simulations. The numbers represent seedlings per hectare.

<table>
<thead>
<tr>
<th>Management Scenario</th>
<th>Acer saccharum</th>
<th>Fagus grandifolia</th>
<th>Tsuga canadensis</th>
<th>Picea rubens</th>
<th>Fraxinus americana</th>
<th>Betula alleghaniensis</th>
<th>Acer rubrum</th>
<th>Populus tremuloides</th>
<th>Betula papyrifera</th>
</tr>
</thead>
<tbody>
<tr>
<td>Clearcut</td>
<td>4448</td>
<td>1730</td>
<td>432</td>
<td>432</td>
<td>8154</td>
<td>8093</td>
<td>8093</td>
<td>15320</td>
<td>15320</td>
</tr>
<tr>
<td>Shelterwood</td>
<td>4448</td>
<td>4695</td>
<td>62</td>
<td>62</td>
<td>618</td>
<td>556</td>
<td>1174</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>ITS_Low Intensity</td>
<td>2471</td>
<td>1730</td>
<td>309</td>
<td>309</td>
<td>62</td>
<td>62</td>
<td>185</td>
<td>-</td>
<td>62</td>
</tr>
<tr>
<td>ITS_High Intensity</td>
<td>1977</td>
<td>2224</td>
<td>309</td>
<td>309</td>
<td>62</td>
<td>57</td>
<td>185</td>
<td>-</td>
<td>62</td>
</tr>
<tr>
<td>Background</td>
<td>494</td>
<td>247</td>
<td>62</td>
<td>62</td>
<td>-</td>
<td>62</td>
<td>62</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>
Table 8: Description of independent variables used in CART analysis. The character of variables is denoted by A = Silvicultural scenario, S = Spatial, E = Environmental, C = Stand composition, T = Stand structure; and the type by N = numeric, O = ordinal, or C = categorical

<table>
<thead>
<tr>
<th>Variable</th>
<th>Character</th>
<th>Type</th>
<th>Values</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Scenario Code</td>
<td>A</td>
<td>C</td>
<td>A - I</td>
<td>A (Background), B (ITS_HighHigh), C (ITS_LowHigh), D (ITS_HighHigh), E (ITS_LowHigh), F (Clearcut_Low), G (Clearcut_High), H (Shelterwood_Low), I (Shelterwood_High)</td>
</tr>
<tr>
<td>Eco-subregion</td>
<td>S</td>
<td>C</td>
<td>10</td>
<td>Ecological subregions as defined by the USDA, 2005, Forest Service ECOMAP team, Washington D.C.</td>
</tr>
<tr>
<td>Site Index</td>
<td>E</td>
<td>N</td>
<td>30 &lt; x &lt; 90</td>
<td>Site index for sugar maple at tree age 50</td>
</tr>
<tr>
<td>Aspect</td>
<td>E</td>
<td>N</td>
<td>0 &lt; x &lt; 359</td>
<td>Aspect in degrees for individual stands</td>
</tr>
<tr>
<td>Percent Conifer</td>
<td>C</td>
<td>N</td>
<td>0 &lt; x &lt; 63</td>
<td>Starting percent conifer, calculated as a percentage of basal area per hectare</td>
</tr>
<tr>
<td>Basal Area</td>
<td>T</td>
<td>N</td>
<td>24 &lt; x &lt; 49</td>
<td>Starting basal area (m²/ha),</td>
</tr>
<tr>
<td>Quadratic Mean Diameter</td>
<td>T</td>
<td>N</td>
<td>1.8 = x = 4.5</td>
<td>Starting QMD. QMD is the diameter of the tree of average basal area.</td>
</tr>
<tr>
<td>Structure Class</td>
<td>T</td>
<td>O</td>
<td>0 - 6</td>
<td>0 (bare ground), 1 (stand initiation), 2 (stem exclusion), 3 (understory reinitiating), 4 (young forest, multi-strata), 5 (old forest, single stratum), 6 (old forest, multi-strata) (Crookston and Stage 1999)</td>
</tr>
<tr>
<td>Number of strata</td>
<td>T</td>
<td>O</td>
<td>0 - 3</td>
<td>Strata differentiated by 30% differentiation in tree height, with minimum threshold of 5% cover to qualify as a strata (Crookston and Stage 1999)</td>
</tr>
<tr>
<td>Slope</td>
<td>E</td>
<td>N</td>
<td>0 - 30</td>
<td>Percent slope steepness for individual stands</td>
</tr>
<tr>
<td>Stand age</td>
<td>T</td>
<td>N</td>
<td>80 = x = 100</td>
<td>Starting stand age</td>
</tr>
</tbody>
</table>
Table 9: Mean C storage over the 160 year simulation period for several different pools (N=32).

<table>
<thead>
<tr>
<th>Management Scenario</th>
<th>Total C with Wood Products (Mg C/ha)</th>
<th>Aboveground Live (Mg C/ha)</th>
<th>Standing Dead (Mg C/ha)</th>
<th>Down Dead Wood Products (Mg C/ha)</th>
<th>Landfill (Mg C/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>No Management</td>
<td>157 ± 9</td>
<td>140 ± 8</td>
<td>7 ± 0.5</td>
<td>13 ± 1</td>
<td>0 ± 0</td>
</tr>
<tr>
<td>ITS_HighLow</td>
<td>113 ± 5</td>
<td>83 ± 3</td>
<td>0.6 ± 0.2</td>
<td>9 ± 1</td>
<td>9 ± 1</td>
</tr>
<tr>
<td>ITS_HighHigh</td>
<td>107 ± 5</td>
<td>75 ± 3</td>
<td>0.3 ± 0.1</td>
<td>9 ± 1</td>
<td>10 ± 1</td>
</tr>
<tr>
<td>ITS_LowLow</td>
<td>98 ± 5</td>
<td>63 ± 2</td>
<td>0.3 ± 0.1</td>
<td>8 ± 1</td>
<td>11 ± 1</td>
</tr>
<tr>
<td>ITS_LowHigh</td>
<td>91 ± 4</td>
<td>54 ± 2</td>
<td>0.2 ± 0.04</td>
<td>9 ± 1</td>
<td>12 ± 1</td>
</tr>
<tr>
<td>Shelterwood_Low</td>
<td>90 ± 5</td>
<td>64 ± 5</td>
<td>0.2 ± 0.1</td>
<td>7 ± 0.4</td>
<td>9 ± 1</td>
</tr>
<tr>
<td>Shelterwood_High</td>
<td>90 ± 5</td>
<td>65 ± 4</td>
<td>0.2 ± 0.1</td>
<td>7 ± 0.4</td>
<td>8 ± 1</td>
</tr>
<tr>
<td>Clearcut_Low</td>
<td>74 ± 5</td>
<td>31 ± 3</td>
<td>0.1 ± 0.03</td>
<td>9 ± 1</td>
<td>17 ± 1</td>
</tr>
<tr>
<td>Clearcut_High</td>
<td>72 ± 5</td>
<td>29 ± 3</td>
<td>0.1 ± 0.04</td>
<td>10 ± 1</td>
<td>15 ± 1</td>
</tr>
</tbody>
</table>
Table 10: Treatment effects on the mean C sequestration over the 160 year simulation period, based on two-way ANOVA. Italicized p values are statistically significant.

<table>
<thead>
<tr>
<th>Treatment</th>
<th>Silviculture type</th>
<th>Mean Square</th>
<th>F</th>
<th>Significance (p)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Harvesting Frequency* Retention (interaction)</td>
<td>Total</td>
<td>92.1</td>
<td>.300</td>
<td>.584</td>
</tr>
<tr>
<td></td>
<td>Even-age</td>
<td>71.1</td>
<td>.352</td>
<td>.554</td>
</tr>
<tr>
<td></td>
<td>Uneven-age</td>
<td>26.4</td>
<td>.133</td>
<td>.716</td>
</tr>
<tr>
<td>Harvesting Frequency</td>
<td>Total</td>
<td>940.2</td>
<td>3.07</td>
<td>.081</td>
</tr>
<tr>
<td></td>
<td>Even-age</td>
<td>39.8</td>
<td>.197</td>
<td>.658</td>
</tr>
<tr>
<td></td>
<td>Uneven-age</td>
<td>1373.4</td>
<td>6.91</td>
<td>.010</td>
</tr>
<tr>
<td>Retention</td>
<td>Total</td>
<td>17575.9</td>
<td>57.3</td>
<td>.000</td>
</tr>
<tr>
<td></td>
<td>Even-age</td>
<td>9674.5</td>
<td>48.0</td>
<td>.000</td>
</tr>
<tr>
<td></td>
<td>Uneven-age</td>
<td>7944.0</td>
<td>40.0</td>
<td>.000</td>
</tr>
</tbody>
</table>
Table 11: Two-way ANOVA results from sensitivity analysis. Results are divided by harvesting frequency and structural retention. Harvesting frequency adjustments are shown as percent above (+) or below (-) the original high and low harvesting frequencies used in simulation modeling. Four harvesting frequencies were used: 1) 25% below the original high frequency (60 years even-age; 11 years uneven-age); 2) the original high frequency (80 years even-age; 15 years uneven-age); 3) the original low frequency (120 years even-age; 30 years uneven-age); 4) 25% above original low frequency (150 years even-age; 38 years uneven-age). Italicized p values are statistically significant.

<table>
<thead>
<tr>
<th>Treatment</th>
<th>Silviculture type</th>
<th>Harvesting Frequency</th>
<th>Mean Square</th>
<th>F</th>
<th>Significance (p)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Harvesting Frequency*</td>
<td>Even-age</td>
<td>- 25%</td>
<td>14955.3</td>
<td>94.7</td>
<td>.000</td>
</tr>
<tr>
<td>Retention (interaction)</td>
<td></td>
<td>+/- 25%</td>
<td>17339.0</td>
<td>103.4</td>
<td>.000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>No change</td>
<td>71.1</td>
<td>.352</td>
<td>.554</td>
</tr>
<tr>
<td></td>
<td></td>
<td>+ 25%</td>
<td>317.4</td>
<td>1.50</td>
<td>.223</td>
</tr>
<tr>
<td></td>
<td>Uneven-age</td>
<td>- 25% *</td>
<td>67.8</td>
<td>.326</td>
<td>.569</td>
</tr>
<tr>
<td></td>
<td></td>
<td>+/- 25% *</td>
<td>67.8</td>
<td>.326</td>
<td>.569</td>
</tr>
<tr>
<td></td>
<td></td>
<td>No change</td>
<td>26.4</td>
<td>.133</td>
<td>.716</td>
</tr>
<tr>
<td></td>
<td></td>
<td>+ 25%</td>
<td>67.8</td>
<td>.326</td>
<td>.569</td>
</tr>
<tr>
<td>Harvesting Frequency</td>
<td>Even-age</td>
<td>- 25%</td>
<td>17935.0</td>
<td>113.6</td>
<td>.000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>+/- 25%</td>
<td>29779.8</td>
<td>177.6</td>
<td>.000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>No change</td>
<td>40.0</td>
<td>.197</td>
<td>.658</td>
</tr>
<tr>
<td></td>
<td></td>
<td>+ 25%</td>
<td>2020.6</td>
<td>9.56</td>
<td>.002</td>
</tr>
<tr>
<td></td>
<td>Uneven-age</td>
<td>- 25% *</td>
<td>3811.7</td>
<td>18.4</td>
<td>.000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>+/- 25% *</td>
<td>3811.7</td>
<td>18.4</td>
<td>.000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>No change</td>
<td>1373.4</td>
<td>6.90</td>
<td>.010</td>
</tr>
<tr>
<td></td>
<td></td>
<td>+ 25%</td>
<td>3811.7</td>
<td>18.4</td>
<td>.000</td>
</tr>
<tr>
<td>Retention</td>
<td>Even-age</td>
<td>- 25%</td>
<td>45037.8</td>
<td>285.2</td>
<td>.000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>+/- 25%</td>
<td>41142.1</td>
<td>245.4</td>
<td>.000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>No change</td>
<td>9674.5</td>
<td>48.0</td>
<td>.000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>+ 25%</td>
<td>7916.2</td>
<td>37.4</td>
<td>.000</td>
</tr>
<tr>
<td></td>
<td>Uneven-age</td>
<td>- 25% *</td>
<td>7402.1</td>
<td>35.6</td>
<td>.000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>+/- 25% *</td>
<td>7402.1</td>
<td>35.6</td>
<td>.000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>No change</td>
<td>7944.0</td>
<td>40.0</td>
<td>.000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>+ 25%</td>
<td>7402.1</td>
<td>35.6</td>
<td>.000</td>
</tr>
</tbody>
</table>

Note: * = As a result of model limitations, 11 year harvesting frequencies in uneven-aged scenarios are simulated the same as 15 year entry cycles and values are identical.
Table 12: Comparison of three different calculated mean C uptake rates by management scenario.

<table>
<thead>
<tr>
<th>Management Scenario</th>
<th>Harvesting Frequency (years)</th>
<th>Forest C uptake rate per harvesting cycle (Mg C·ha(^{-1})·yr(^{-1}))</th>
<th>Forest C uptake rate for 160 year simulation period (Mg C·ha(^{-1})·yr(^{-1}))</th>
<th>Forest and harvested wood products C uptake rate for 160 year simulation period (Mg C·ha(^{-1})·yr(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>Clearcut_High</td>
<td>80</td>
<td>0.55 ± 0.05</td>
<td>0.23 ± 0.03</td>
<td>0.23 ± 0.05</td>
</tr>
<tr>
<td>Clearcut_Low</td>
<td>120</td>
<td>0.44 ± 0.05</td>
<td>0.02 ± 0.03</td>
<td>0.08 ± 0.05</td>
</tr>
<tr>
<td>Shelterwood_High</td>
<td>80</td>
<td>0.18 ± 0.05</td>
<td>0.13 ± 0.02</td>
<td>0.13 ± 0.03</td>
</tr>
<tr>
<td>Shelterwood_Low</td>
<td>120</td>
<td>0.17 ± 0.04</td>
<td>-0.02 ± 0.02</td>
<td>0.02 ± 0.03</td>
</tr>
<tr>
<td>ITS_LowHigh</td>
<td>15</td>
<td>-0.02 ± 0.02</td>
<td>-0.04 ± 0.01</td>
<td>0.07 ± 0.03</td>
</tr>
<tr>
<td>ITS_LowLow</td>
<td>30</td>
<td>-0.01 ± 0.02</td>
<td>-0.04 ± 0.01</td>
<td>0.08 ± 0.03</td>
</tr>
<tr>
<td>ITS_HighHigh</td>
<td>15</td>
<td>0.04 ± 0.03</td>
<td>0.02 ± 0.02</td>
<td>0.14 ± 0.09</td>
</tr>
<tr>
<td>ITS_HighLow</td>
<td>30</td>
<td>0.05 ± 0.02</td>
<td>0.02 ± 0.02</td>
<td>0.14 ± 0.09</td>
</tr>
<tr>
<td>No Management</td>
<td>NA</td>
<td>0.36 ± 0.04</td>
<td>0.36 ± 0.04</td>
<td>NA</td>
</tr>
</tbody>
</table>
2.9 Figures

Figure 6: Map of approximate locations of FIA plots used in simulation modeling. In total, we selected 32 stands spanning 10 eco-subregions and 4 states.
Figure 7: Simulation output time series for the 9 different management scenarios (values represent 10 year mean C storage of 32 stands). Ten year means of C sequestration were used to create chronosequences to illustrate the temporal dynamics for each management scenario. For management scenario descriptions refer to tables 2 and 3.
Figure 8: Comparison of mean C stocks in nine different management scenarios. Chronosequences starts immediately following the first harvest in 2005. Error bars show + one standard error of the mean. For management scenario descriptions refer to tables 2 and 3.
Figure 9: Classification and regression tree (CART) showing independent variables selected, split values, and partitioned mean values (bottom) of the dependent variable (mean C sequestration). The figure ranks independent variables by predictive strength (top to bottom); the length of each vertical line is proportional to the amount of deviance explained by each variable. Independent variables were selected from an initial set of 11 variables. Minimum observations required for each split = 5; minimum deviance = 0.05; N = 288.
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