

ABSTRACT

Title of dissertation: **INTEGRATING FOREST ECOLOGICAL
PROCESSES, MANAGEMENT, AND
CARBON PAYMENTS TO ASSESS
ECOSYSTEM SERVICE TRADE-OFFS**

Pui-Yu Ling, Doctor of Philosophy, 2016

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Due to the current crisis on global warming and species extinction, scientists and economists have proposed to quantify ecosystem goods and services to establish policies that can preserve the ecosystem, provide better livelihood to the local stakeholders, and benefit society. The overarching question for this dissertation study is, “How to quantify the trade-offs between carbon sequestration, total timber harvested, and habitat provision for sustaining biodiversity?” In particular, when i. different management regimes; ii. carbon prices; and iii. natural disturbances; interact with each other. The first step is to develop a model that quantifies the relationship between forest management activities and the amount of timber harvested annually. To provide a general understanding of the relationship, the study area for the first step is at the county level for the whole state of North Carolina. The following 2 steps involve more specific studies to include the interaction between management activities, natural disturbances, species competition during the forest

succession process. Therefore, a species-rich, heterogeneous area with an active timber industry, located in western part of North Carolina, the Grandfather Ranger District was chosen. The second step is to quantify the outcomes and analyze the trade-offs between the ecosystem services under different management scenarios, coupled with the influences from natural disturbance. The third step combines the result from the second step with different carbon prices and interest rates to obtain a fine scale and spatial analysis of the resulting revenue.

The dissertation is the first step of research in developing a dynamic model that fully couples the social and forest ecological system. Besides the potential uses for fulfilling the requirements of carbon accounting, the result from the first step of the dissertation is the maps of annual volume of harvested timber of various types. The maps can be used to further study to analyze the relationship between the amount of timber harvested as a result of different policies and prices at different places. The second step demonstrated a way to quantify the trade-offs between the selected ecosystem services under different forest management regimes and influences from natural disturbances events. Maps of revenue from both selling the carbon credit and harvested timber for scenarios of different carbon prices and interest rates were produced in the last step. The choice of future timber production as a result of carbon and roundwood prices can partly be understood by utilizing the maps from the first step in this dissertation. If the factors affect individual choices of harvest at the local level and those affect the supply of harvested timber at the regional level are understood, then a dynamic model that consists of management decisions, natural

disturbance, ecosystem service valuation policies, and forest ecosystem response can be developed. Such model can help better understand the ecosystem services valuation policy on the financial well-being of foresters and the health of the forest ecosystem.

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by

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Preface

This dissertation includes an introductory chapter, three chapters describing the research, and a conclusion chapter. Chapter 2 was coauthored with Giovanni Baiocchi and Chengquan Huang and has been published in the journal *Climatic Change*; chapter 3 was coauthored with Stephen Prince, Caren Dymond, Wemin Xi, George Hurtt, and Giovanni Baiocchi and is prepared to submit to the *Journal of Applied Ecology*; chapter 4 is on-going research that will be submitted for publication.

Dedication

To my parents: Ms. LEUNG Man Wah and Mr. LING Yue Wah

AND

To my aunt & uncle: Ms. LING Yuen Yee and Mr. HUI Chan Sun

AND

To our mother Earth

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List of Abbreviations

α	alpha
β	beta
VCT	Vegetation Change Tracker
IPCC	Intergovernmental Panel on Climate Change
USFS	United States National Forest Service
NLCD	National Land Cover Database
REDD+	Reduced Emissions through Avoided Deforestation and Forest Degradation
NIPF	Non Industrial Private Forest
HWP	Harvested Wood Product

Chapter 1: Introduction

1.1 What are the dynamics between the ecological and economic systems?

With the pressure on increasing human population, conservation efforts have shifted beyond merely preservation into encompassing humans' need and demand on ecosystem services provision (Daily & Matson, 2008). Human utilization of natural resources clearly has effects on the natural processes that produce them, so understanding the interaction between the ecosystem and the economic system is one of the most important elements in designing policies on sustainability (Carpenter et al., 2009). While resource management is generally practiced in sustainable systems, and can be quantified in financial terms, other components often cannot and are therefore not considered (Nelson & Daily, 2010). For example, commercial forestry depends on the natural process of tree growth. This is followed by industrial processing to make a saleable product, the price of which can be known with high precision. The only other costs for the foresters are harvest, processing, and management of the forest for future harvests. However, the costs to society of loss of carbon fixation, effects on biodiversity and recreational use of the forest, for exam-

ple, are not quantified, in fact cannot easily be quantified in most cases (Daily et al., 2000). Those factors that do not have market values, however, are currently being considered in policy making and these may have important effects on the cycle from forest growth, through harvest and sale back to forest management to sustain the supply of marketable wood products. These policies include, for example, subsidies for carbon sequestration paid to forestry enterprises (which can be quantified monetarily), creation of conservation reserves, tourism, and policies intended to preserve biodiversity – which cannot be easily quantified. Furthermore, these actions may induce feedbacks and even instabilities.

1.2 Objectives

The specific forest ecosystem services focused in this dissertation study are carbon sequestration, roundwood production, and habitat provision for sustaining biodiversity. The overarching question for this dissertation study is, “How to quantify the trade-offs between carbon sequestration, total timber production, and habitat provision for sustaining biodiversity?” In particular, when i. different management regimes; ii. carbon prices; and iii. natural disturbances; interact with each other. Since each step can be complex, process and economic models will be adapted and linked to enable the effects of subsidies, roundwood price, and forest disturbances (see figure 1.1 for details). Specifically:

1. How do forest management activities contribute to the amount of harvested wood products?
2. How to quantify the contribution of forest management activities to the carbon storage in roundwood products?
3. How are the carbon sequestration and species composition impacted by natural disturbances and various management regimes in a species-rich environment?
4. How do different management regimes affect the trade-offs between carbon sequestration, roundwood production, and habitat provision for sustainability?
5. How do different carbon credits and interest rates affect the income of foresters spatially?
6. Which management regime would the foresters likely to adopt if they were to maximize their profit?

1.2.1 Definition of terms in context

- Ecosystem services: The Millennium Ecosystem Assessment report defined “ecosystem services” as “the benefits people obtain from ecosystems” and classified four types of services that the ecosystems provide to society ([Millennium Ecosystem Assessment, 2005](#)). A lot of those services do not have commercial values or traded in markets. There is a trade-off between some

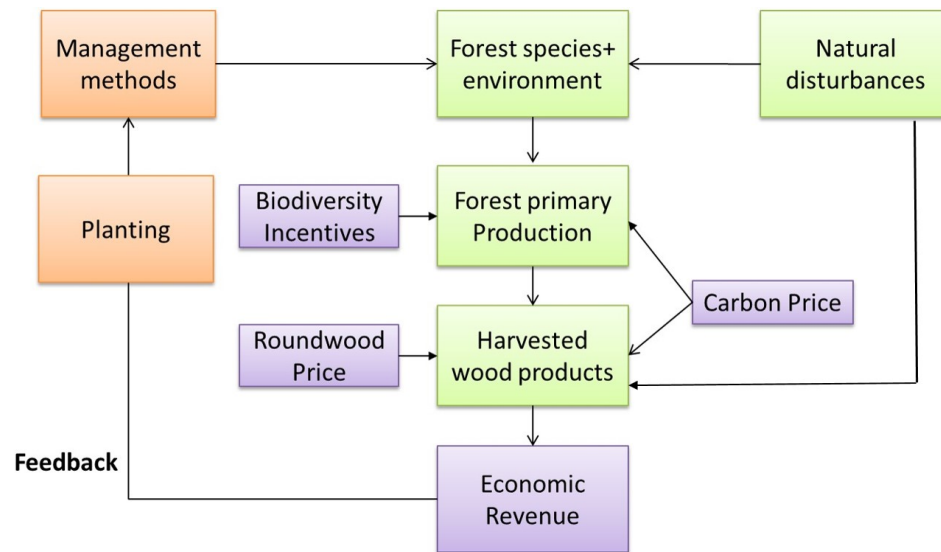


Figure 1.1: Conceptual diagram of the interaction between the forest ecological and economic systems. The green boxes are the interaction within the natural system and its resulting ecosystem goods (commercial products). The purple boxes indicate the external forces from the government and the market that would influence the economic revenue of the landowners. The red boxes show the actions of the landowners that may be influenced as the result of the economic revenue they received, which will in turn influence the forest ecology.

ecosystem services: a policy that favors one may cause an unintentional loss of another.

- Ecosystem resilience: It is the ability of the ecosystem to return to its similar structure and functioning before disturbances. Under the influences of climate change and human intervention, a forest may not be able to recover and to provide the same ecosystem services as before the disturbance events ([Reyer et al., 2015](#)).
- Feedback and interaction between ecological and economic systems: Forest policies and management practices influence the provision of forest ecosystem services, which would influence the income of the foresters. Foresters in turn may change their management practices.

1.2.2 Status of current research and scientific motivation

A spatial model that includes forest succession, species competition, management regimes, and natural disturbances allows more realistic simulation of the impact of the forest landowners' reaction as a result of ecosystem services valuation policy. When there is an additional source of income based on carbon sequestration, foresters are going to alter their management regimes if their goal is to maximize their profit. The alternation in the management regimes will affect the forest composition and structure. Hence, the suitability of a forest in providing habitat to sustain biodiversity. Currently, there is not enough understanding of the feedback

between the impact of ecosystem valuation policies on ecosystem services provision and the ecosystem structure, especially with the influence from climate change and natural disturbances ([Guerry et al., 2015](#)).

A complex mechanistic forest succession model is needed to understand especially the impact of natural and anthropogenic disturbances on “biodiversity”. There has been a debate between ecologist and economist when trying to value “biodiversity” (see [Bartkowski et al. \(2015\)](#) and [Farnsworth et al. \(2015\)](#) for example), as the meaning and definition of this term is unclear. Research often lacks sufficient components for evaluating “biodiversity”. A complex forest succession model that includes species interaction can provide more comprehensive way to value ecosystem services that relate to the quality of habitat. Previously studies of this system have been highly simplified, for examples, the studies by [Touza et al. \(2008\)](#), [Niimäki et al. \(2013\)](#), and [Mäkipää et al. \(2014\)](#), are not spatial and do not include species competition and forest succession, or the influences of natural disturbances, such as wind and fire. Even spatially explicit tools, such as InVEST ([Tallis & Polasky, 2011](#)), only considers a simple growth and yield model in understanding the trade-offs between timber production and carbon sequestration of forest ecosystem. These models and tools are only useful for making general policies, but to investigate the impact of forest management on the interaction of tree species, a more complex model is needed ([Tallis & Polasky, 2011](#)).

1.2.3 Scale of the study

Identification of scales, the stakeholders, and especially to whom the benefits of the systems services accrue, allows the analysis of potential conflicts in environmental management (Hein et al., 2006; Spies & Johnson, 2003). Currently, there are two approaches that have been used for generating the general ecosystem service assessments at scales that are meant to influence policy decisions. The first one is a broad-scale assessment of multiple services. It extrapolates the value estimation per unit area by biome to entire regions or planet based on habitat types. The value is estimated by summing the value of marketed ecosystem services and the estimation of the willingness-to-pay of individuals for non-marketed ecosystem services (Costanza et al., 1997). However, it does not allow for analyses of service provision and changes in value under new conditions. The second one is a fine-scale assessment of a single service in a small area with an ecological production function that depends on local ecological variables. However, it lacks both the scope and scale that are relevant to policy decision making (Nelson et al., 2009). In addition, the analysis may be overwhelmed by the information volume at a fine level of details (Spies & Johnson, 2003). To combine the rigor of the fine-scale studies with the breath of the broad-scale assessment, a scale that includes all the studied ecosystem goods and services and environmental management activities is the optimal spatial scale in coupling research on ecological processes with ecosystem services valuation (De Groot et al., 2010; Müller et al., 2010). The physical scale of the ecosystem

function and the scale at which humans value the goods and services provided are not necessarily the same (De Groot et al., 2002; Hein et al., 2006). In the case of forestry, the non-industrial private forest (NIPF) and industrial forest owners typically lack the incentive to manage their land to provide ecosystem services, such as carbon sequestration, because the benefits are public goods (Nelson et al., 2008). For the valuation of carbon sequestration, the beneficiaries are at a higher, global level, the payments of subsidies will occur on the government level to make it more profitable for the landowners to manage their forest to sequester more carbon (Nelson et al., 2009). Environmental management activities such as timber harvesting, and the broad-range effects such as natural disturbances on a forest, are mostly concentrated at the landscape scale (Müller et al., 2010; Scheller & Mladenoff, 2007). Landscape ecology can be used to address this problem because it explicitly links ecological processes and management (Moorcroft et al., 2001).

1.2.4 Study area

Like most of the Eastern forests, North Carolina (NC) forests nowadays are intensively managed to provide goods and services for humans (Brown & New, 2012). But unlike other states, several attributes of North Carolina favors the successful implementation of forest-based carbon sequestration policy: productive soil, abundant privately-owned working forest, diverse forest species composition, accessible to investment capital to support forestry activities and a large forestry-related economy (North Carolina Division of Forest Resources, 2010). In addition to the availabil-

ity of Landsat-derived forest disturbance data ([Huang et al., 2015](#))), these factors provide an opportunity to test the dynamic between forest landowners management decision, carbon dynamics, and the trade-offs between timber products and carbon credit in North Carolina.

1.2.5 Dissertation outline

The work of my dissertation is organized in three parts, from Chapters 2 to 4, as follows: The goal of chapter 2 is to develop a quantifiable link between forest management activities and different types of roundwood harvested. An empirical model was developed that links between forest management activities and annual carbon flux from forest biomass to the harvested wood product pool from 1986 to 2010. This study utilized time series of historical forest disturbance maps obtained from the Landsat time series stack ([Huang et al., 2010](#)), a forest-type classification map obtained from the NC GAP study ([McKerrow et al., 2006](#)), and the timber product output survey that provides county-level non-continuous time period of the timber (roundwood) harvested ([U.S. Department of Agriculture, 2015](#)). The study in this chapter is the first step in developing a roundwood production function that relates to forest management activities. Since the study does not involve any detail mechanistic process, the study area is at the county level for the whole state of NC (figure [1.2](#)).

The goal of chapter 3 is to assess the impact of forest ecosystem services as the result of different forest management regimes and influences from natural dis-

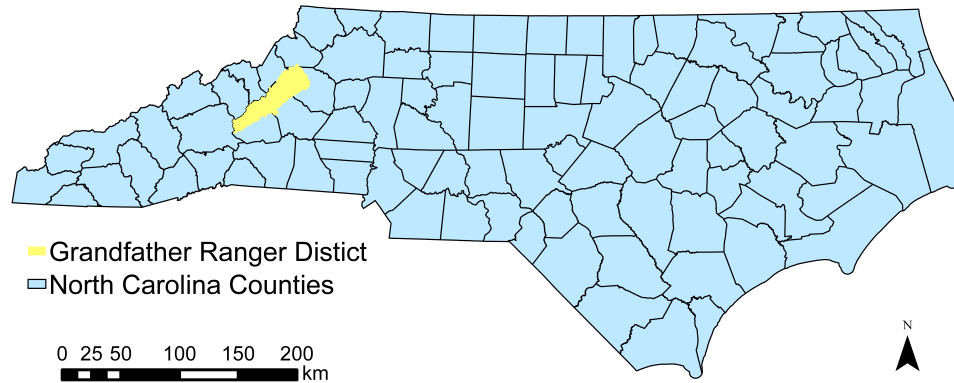


Figure 1.2: The state of North Carolina is the study area of Chapter 2 of the dissertation. The Grandfather Ranger District (GRD), highlighted in yellow, is the study area of Chapters 3 and 4.

turbance. Specific forest ecosystem services for the study are: carbon sequestration, roundwood output, and habitat provision for sustainability, which is a function of the forest species composition. The study utilized a spatially explicit, landscape-scale forest succession model, Landis-II (Scheller & Mladenoff, 2007), to include the complex forest succession processes in simulating the possible future outcome for different management scenarios. Because the study involves detail forest succession processes and species interaction, as well as natural disturbances, the area of study is the Grandfather Ranger District (GRD) located in western North Carolina (figure 1.2, area highlighted in yellow).

The goal of chapter 4 is to analyze the impact of different carbon credits and interest rates on the income of different forest areas with different management goals for different management scenarios. This study combines the result from chapter 3 to obtain a fine scale analyze of the resulting income as a result of different carbon pricing. Since the result at such fine scale is the most relevant to the local

stakeholders, it can be used to analyze the possible impact and hence, to deduce the possible future management regimes applied.

Chapter 5 concludes the finding of this dissertation, its implications, and possible further developments of the research completed in this dissertation.

Chapter 2: Estimating Annual Influx of Carbon to Harvested Wood Products Linked to Forest Management Activities Using Remote Sensing

Abstract A new framework, based on Landsat time series data and forest inventories, was developed to estimate the carbon in roundwood harvested from forest management activities, which will enter the HWP pool and remain stored in end uses and landfills. The approach keeps the distinction between the carbon from different types of roundwood sources, which allows for better integration with the regional HWP carbon lifetime information. Existing methods that are based on large scale regional/national values and linear interpolation of data gaps, can provide only very approximate carbon estimates. The model was applied to a US state using county level data, but can also suit different areas as long as sufficient harvest records are available for calibration. The results can be used to study managed forests and evaluate the impact of forest policies on the carbon cycle at a detailed scale. The estimated quantity of carbon in roundwood harvest provides an upper bound on the gross carbon added to HWP in use, prior to deductions from losses. The results can also be coupled with mill processing efficiency estimate and wood product life

cycle analysis to better understand the effect of forest management activities on the carbon cycle.

2.1 Introduction

There is a growing recognition in the carbon accounting and climate policy literature of the importance of harvested wood products (HWP) as a carbon sink as well as substitutes for some of the fossil-intensive products. HWP has also been proposed as a means to store carbon (Lippke et al., 2011; Ruddell et al., 2007; Zeng, 2008). Ellison et al. (2011) proposed a post-Kyoto framework that includes both conserving forests and promoting the use of HWP as a carbon sink, so that timber-rich countries, whether developing or developed, can come to an agreement of a carbon trading and crediting system that can both preserve forests and recognize the importance of carbon stored in HWP. Along the same lines, the most recent IPCC report acknowledges the tradeoffs between the importance of forest as a carbon sink, its importance in maintaining ecological diversity and the reservoir role of HWP (Smith & Bustamante, 2014, chapter 11).

The newly released IPCC *Guideline* for estimating anthropogenic greenhouse gas (GHG) emissions due to land-use and cover changes encourages the adoption of best practice in producing transparent and verifiable data of complete annual time series and detailed spatial coverage of the impact of forest management activities and natural disturbances on the GHG fluxes (IPCC, 2014, section 2.2.2–2.2.4). It also sets out provisions for estimating the share of HWP originating from domestic

forests under different management regimes¹. The major steps are: estimate the fraction of roundwood harvest in a forest management activity; estimate the fraction of the roundwood (industrial roundwood²) harvest that will serve as the feedstock of HWP; and finally, break down the fraction of roundwood products into different primary wood products in-use or in land fill using regional specific look-up tables (IPCC, 2014, section 2.8.1.2).

The United States, also provides similar guidelines for forest landowners to report the contribution of their harvest to the HWP. The latest USDA guidelines (Hoover et al., 2014) state that the estimation can start from the wood products in-use (Skog et al., 2004; Smith et al., 2006) or, alternatively, from roundwood production data (Smith et al., 2006). Relevant lookup tables can then be used by forest landowners to estimate the fate of the carbon stored after harvest over the next 100 years (Hoover et al., 2014). This approach requires reliable spatial and temporal information on roundwood production for its implementation. Current approaches to estimating the carbon stored in the HWP provide estimates either at a national or regional level (see, e.g., Apps et al., 1999; Donlan et al., 2012; Heath et al., 2011; Karjalainen et al., 1994; Masek et al., 2011; Pan et al., 2011; Stockmann et al., 2012; Zhou et al., 2013), or for specific forests (Profft et al., 2009)

¹on Afforested or Reforested lands; harvest yield from Deforested lands is excluded (section 2.5 and section 2.6 in the IPCC report(IPCC, 2014))

²Roundwood is further divided in industrial and non-industrial. Non-industrial roundwood is fuelwood that is assumed to be burned within a year after harvest. Only industrial roundwood serves as a feedstock to HWP.

over a few years. Other studies do not provide the contribution of forest management activities (Adams et al., 2006) or detailed information on the types of roundwood produced (Huang et al., 2015) which is needed for further analysis of the dynamics of roundwood production and life cycle analysis, to improve accounting of long-term carbon stored.

This chapter proposes a method of estimating the annual influx of carbon to different types of roundwood from forest management activities³ by integrating remote sensing with the limited forest inventory data on timber production. Results can serve as a starting point for determining the fraction of carbon in HWP that remains in end-use and in landfill by using specialized look-up tables, e.g. Smith et al. (2006), or more specific life cycle assessments. The proposed framework provides a direct link between forest management activities and carbon flux to the HWP pool that can help implement the latest IPCC guidelines into practice (IPCC, 2014, p. O10). Spatially linking roundwood harvest with forest activities in domestic forests constitutes the first step in estimating the fraction of HWP contribution from each forest activity. Particularly the one from forest land conversion to other land uses (deforestation), as carbon stored in HWP resulting from deforestation is assumed to oxidize instantaneously according to the IPCC guidelines, and thus has to be excluded. To fulfill the specific definitions of deforestation stated in section 2.6 of the Guidelines (IPCC, 2014), additional detailed land-use classification informa-

³Forest management activities, such as harvest and salvage logging, which contribute to roundwood production, are observed as forest disturbance events in remotely sensed images.

tion is needed to exclude the HWP contribution from deforestation in the proposed framework. The next section describes the approach in more details.

2.2 Methods

2.2.1 Model framework

This study links the carbon in the HWP with forest disturbance areas detected from remote sensing images. The estimation method provides an upper bound of the carbon that enters the HWP pool, because some roundwood is left in the mill during processing. The current methods use field survey-based inventories of roundwood but these are not available every year. In the proposed method, missing years of production are estimated using remotely sensed forest disturbance maps, calibrated with survey data from the forest inventory. Often survey based inventories of roundwood produced are not available every year. Forest disturbance maps showing forest changes with the associated intensities, which is directly related to biomass change, are updated annually from 1985 to present for the continental United States and Canada. These maps are derived from Landsat time series stack using the Vegetation Change Tracker (VCT) algorithm by [Huang et al. \(2010\)](#). In areas where timber harvest is the major cause of forest change, the maps can be used to estimate the time series of different roundwood types production. The area of disturbed forest with its associated intensity are partitioned between softwood, hardwood, or mixed forest disturbance by using a land cover classification map. Hardwood and softwood

forests disturbed areas are used to estimate the corresponding type of roundwood production, while mixed forest disturbed areas are used to estimate both the hardwood and softwood roundwood production. The amount of each type of roundwood produced at the county level is converted to mass of carbon equivalent using information provided by [Smith et al. \(2006\)](#).

2.2.2 Statistical model linking roundwood production to forest disturbance

A regression model that captures the unobserved area-specific (e.g., difference in biophysical factors, forest size, climate, and management style) and time-specific characteristics (e.g., prices and meteorological event affecting all areas) was used to link the production of different roundwood types with the observed area of disturbed forest and its associated disturbance intensities (see [Appendix A.1](#) for details on the model selection strategy). The equation of the model can be written as:

$$Y_{it} = \alpha_i + \beta X_{it} + \tau_t + \varepsilon_{it} \quad (2.1)$$

where Y_{it} , in this specific case, is the roundwood production survey data by species and by geographic location (e.g., county), i indexes unit-areas and t the time period. X_{it} denotes the area of forest disturbance in each of the forest type (such as hardwood or softwood or mixed) available and the degree of disturbance intensity in each unit area of the analysis. The index of disturbance intensity is divided into different classes based on its values, ranging from slight thinning to complete clear-cutting.

α_i denote the unit-areas effects, β is the vector of partial effects, τ_t the time specific effects, and ε_{it} the standard random error term. If the α s and τ s are treated as fixed, the model can be estimated by Ordinary Least Squares (OLS) using dummy variables (see [Wooldridge \(2010\)](#), for example, for more details on Panel Data Models). It is assumed that the disturbance areas in each type of forest only contribute to the production roundwood of the corresponding species type and disturbance areas in mixed wood forest contribute to that of both species types. The observed disturbance area for each forest type in each unit area is further partitioned according to the disturbance intensity of the pixel which can distinguish between clear-cutting and different degrees of harvesting. The larger the forest disturbance index value, the more severe the forest disturbance ([Huang et al., 2009](#)).

2.3 Case study

2.3.1 Study area and data used

This model was applied to North Carolina (NC). Located in the Southeastern United States, NC forests are intensely managed for timber harvesting ([Williams et al., 2014](#)). The Forest Service divides NC into four regions: Northern Coastal Plain (NCP), Southern Coastal Plain (SCP), Piedmont, and Mountain. Those four regions match the U.S. Environmental Protection Agency Level III ecoregions ([Griffith et al., 2002](#)), except that the Coastal Plain is divided into North and South. The four regions differ in topography, land use, land ownership, demography, and

tree species ([North Carolina Division of Forest Resources, 2010](#), chap.2). Based on calculation using the data in the Resources Planning Act Assessment reports ([USDA, 2015](#)), NC accounted for about 4.3% of the roundwood production of the continental United States (see [Appendix A.3](#) for details). Like other states in the southeast, most of the forest disturbance occurred can be assumed to be timber harvest.

The first step of this framework is to estimate the annual production of each roundwood type for each county, which serves as the dependent variable of the model (Y_{it}). The types of roundwood production estimated in this study are: sawlog (including veneer), pulpwood, and fuelwood of softwood and hardwood. The amount of roundwood produced in each NC county is available in the Southern Research Station (SRS) Timber Product Output (TPO) report ([U.S. Department of Agriculture, 2015](#)). Except for pulpwood, data on the production of other types of roundwood are not available for every year. The additional pulpwood data will be used to validate the results of the model estimation (see [Appendix A.1.3](#) for details on data availability). Since, in this specific application, the available forest inventory data is completely missing for the years without surveys, the time specific effect of the general model (τ_t) could not be estimated for those years. In addition, for the specific application in this study, the intensity values were divided into four classes based on their distribution.

The independent variables of the model are the annual forest disturbance area with the associated index of disturbance intensity (X_{it}), which are available from

1985 to 2010. Annual forest disturbance area was calculated using the VCT algorithm (Huang et al., 2010). In this model, the annual disturbed forest area is divided into four categories based on the associated pixel-level disturbance intensity calculated by using the *IFZ value*⁴, henceforth, “Magnitude” (Huang et al., 2010, section 3.3.3.2). Forest disturbance events that happened in National Parks were excluded, because they do not yield any roundwood product for sale in the commercial market (Weinberg & Reilly, 2008, chap. 11, sect. 11.04) and are most likely a result of natural disturbances. The disturbed forest area in each category is partitioned into hardwood, softwood, or mixed forest using the GAP Analysis map NC (McKerrow et al., 2006). The land cover map classifies forest areas of different vegetation communities based on the National Land Cover Dataset and local ground surveys (McKerrow et al., 2006). Based on the vegetation community information provided, the dominant vegetation in each community was classified as hardwood or softwood. For forest classes⁵ that both softwood and hardwood as the dominant vegetation, were classified as mixed forest. How the area of the disturbed mixed forest contributes to the production of both hardwood and softwood was determined by an empirical comparison of different model specifications: To select the “best” model, i.e, a model that fits the data well but avoids overfitting it, thus making the model not applicable to other years and areas, a criterion (AIC) that includes a penalty for increasing the number of parameters and two cross-validation procedures based

⁴“IFZ” stands for “integrated forest z-score”, which is a forest index calculated based on the spectral information of the Landsat images, see (Huang et al., 2010) for details.

⁵Wetland forests are excluded.

on leaving sample year and area data out were used (see Appendix [A.1.5](#) for further details).

2.3.2 Calculating roundwood carbon content by type

To obtain the carbon enters the HWP pool, roundwood production, measured in cubic meters, was converted into carbon mass units using the method described in [Smith et al. \(2006\)](#). Volume of roundwood production was converted to dry mass by wood green specific gravity as in [Miles & Smith \(2009\)](#) (See Appendix [A.4](#) for details). Dry mass was converted to carbon equivalent mass using the standard carbon fraction coefficient of 0.5 ([Smith et al., 2006](#)). The unit used for the estimated mass of carbon in roundwood was in kilotonne⁶. The fate of the carbon in HWP will ultimately depend on the type of roundwood and the processing efficiency in mills. The amount of roundwood carbon that is added to HWP in end uses in a given year and will remain in end uses or in landfills in subsequent years can be estimated using tables 6, 8, 9 and D6 in [Smith et al. \(2006\)](#) or tables in [Hoover et al. \(2014\)](#) or other equivalent estimates. Although estimating the permanence of carbon in each type of HWP is outside the scope of this paper, the study results, which provide the link to the spatial and temporal dimensions of forest management activities, can serve as the starting point for a complete life cycle analysis of carbon in HWP.

⁶1 kilotonne = 10^9 grams.

2.3.3 Alternative Approaches of Estimating Annual Production

The result obtained in this study was a time series of carbon mass per year for different types of roundwood based on the production estimates between 1986 and 2009. Two other methods that are employed by the empirical literature to provide estimates of annual time series of roundwood production capable of producing, at least in principle, species details and spatial information, are the linear interpolation of missing values between available data points (see, e.g., [Adams et al., 2006](#); [Bolkesjø et al., 2010](#); [Donlan et al., 2012](#)), and the use of country specific conversion factors between areas affected by harvest and roundwood production volumes, as in [Masek et al. \(2011\)](#). (See [Appendix A.5](#) and discussion section below for more details).

2.3.4 Results

2.3.4.1 Model selection and validation of estimation results

Cross-validation was used to assess the ability of alternate models to accurately predict harvest in years where survey data on harvest is available. Generally, this type of model suffers from variance-bias trade-off since, as the number of variables and interaction terms are increased the residual variance decreases, but such “over fitted” models tend to have low accuracy when applied to different data. Getting the correct balance is crucial for this study as the amount of roundwood harvested

for the years with no production data were estimated. To choose a model that takes both aspects into account, variance and bias can be combined to form the mean squared error (MSE). The model that minimizes MSE was selected, see for example, [Clark & Pregibon \(1993\)](#) by splitting the available sample into 10 equally sized parts (folds). One part was left out and the remaining 9 parts were used for training to estimate the model and tested it on the tenth (see Appendix [A.1.5](#) and table [A.1](#) for details). Plots of the estimated value versus the actual TPO survey value indicated that the final model fitted the data well for each type of roundwood product (fig. [A.1](#)).

2.3.4.2 Comparison with estimates in other studies

The estimated total roundwood production in carbon mass in this chapter was compared with published values from other studies of carbon removed from the live biomass due to forest harvest. Currently, other similar studies are done at the national level. Though downscaling introduces some errors, the results shown in table [2.1](#) gives a rough idea of how the estimates in this study compare with other studies.

table [2.1](#) shows that, at least for this application, other estimates available are not too far off the estimated aggregated values for NC, when properly rescaled, but nevertheless lack details useful to combine with existing HWP carbon life cycle tables (e.g. [Hoover et al., 2014](#); [Smith et al., 2006](#)) to estimate the fate of carbon after removal from the forest.

A map of the average annual carbon influx (upper bound) to all types of HWP between 1986 and 2009 and a map rescaled by the forest area in each county were produced (figure 2.1). These two maps are analogous to those in [Zhou et al. \(2013\)](#), except for the level of details and time span. When forest area is taken into account, the NCP region has a more intense carbon influx to the HWP pool, which agrees with [Zhou et al. \(2013\)](#).

Table 2.1: Comparison between the previously published values and estimates in this study for the annual carbon influx to roundwood through forest harvest (in Mt C/yr); with 95% confidence interval. The “Coverage” column indicates the share of roundwood production for North Carolina (NC) compared to that for the continental US for the corresponding time period. The “Downscale Estimate” column indicates a downscaled value to NC from continental US estimates.

Published Source	US Harvest (Mt C/yr)	Reference Period	Coverage (% of US [†])	Downscale estimate (Mt C/yr)	This estimate (Mt C/yr)
Turner et al. (1995)	124	1980–1989	4.50	5.58	$5.88 \pm 0.11^{\S}$
Heath & Smith (2004)	105	1953–1996	4.50	4.73	$5.73 \pm 0.07^{\P}$
Williams et al. (2012)	107	2005	4.14 [‡]	4.43	4.86 ± 0.13
Zhou et al. (2013)	128	2002–2010	4.14 [‡]	5.29	$4.74 \pm 0.05^{\diamond}$

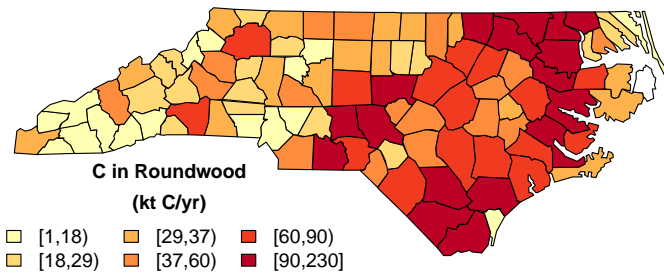
[†]In terms of conterminous US.

[‡]Average percentage for the years 2002 and 2007 were used for downscaling.

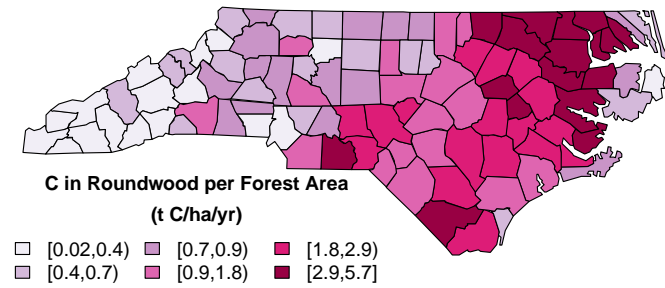
[§]Estimation period is from 1986–1989.

[¶]Estimation period is from 1986–1996.

[◇]Estimation period is from 2002–2009.



(a) Carbon in roundwood



(b) Carbon in roundwood per unit forest area

Figure 2.1: Spatial distribution of (a) the carbon in roundwood products and (b) the carbon in roundwood per unit forest area, averaged over 1986-2009 for each county in North Carolina. Estimates in this study are comparable with the findings in [Zhou et al. \(2013\)](#), but are obtained at a finer spatial scale over longer time period. Maps are generated using the *Maps* ([Brownrigg & Minka, 2011](#)) and the *Classint* ([Bivand et al., 2009](#)) packages in R.

Table 2.2: Carbon in different roundwood types in each ecoregion, averaged over 1986-2009 (in kt C/yr, \pm margin of error, rounded to whole numbers.) The Forest Service divides North Carolina into four regions: Northern Coastal Plain (NCP), Southern Coastal Plain (SCP), Piedmont (Pied.), and Mountain (Mount.). Types of roundwood are: HS=Hardwood Sawlog; HP=Hardwood Pulpwood; HF=Hardwood Fuelwood; SS=Softwood Sawlog; SP=Softwood Pulpwood; SF=Softwood Fuelwood.

Roundwood type	C in Roundwood by Ecoregion (kt C/yr)				
	NCP	SCP	Pied.	Mount.	Whole State
HP	251 \pm 8	325 \pm 11	141 \pm 6	82 \pm 4	799 \pm 16
HS	143 \pm 3	211 \pm 3	312 \pm 5	220 \pm 3	886 \pm 7
HF	107 \pm 2	148 \pm 3	145 \pm 4	89 \pm 3	489 \pm 6
SP	301 \pm 12	474 \pm 18	138 \pm 16	24 \pm 2	936 \pm 28
SS	676 \pm 11	931 \pm 12	461 \pm 11	125 \pm 4	2193 \pm 21
SF	19 \pm 0.3	28 \pm 0.4	13 \pm 0.3	3 \pm 0.1	64 \pm 1
All Types	1498 \pm 19	2117 \pm 25	1210 \pm 21	543 \pm 7	5367 \pm 9

2.3.4.3 Carbon content change by roundwood type over time

The results showed that the annual carbon influx to different roundwood types from forests (figure 2.2) in the Southern Coastal Plain was the highest and the Mountain the lowest (table 2.2).

This is consistent with the species distribution in each region and differences in land ownership⁷ that affect production. Carbon stored in the HWPs that come from sawlog remains in use or in landfill the longest (Smith et al., 2006), which means that, in NC, HWP can be a potential carbon sink. Carbon mass also shows

⁷According to the NC Forest Service (2010): “The Mountains have the highest proportion of publicly owned timberland in the state, mainly because this region includes the Pisgah and Nantahala National Forests. The Mountains have fewer large cities and urban development than the states other regions.”

a decreasing trend over time of the carbon flux to roundwood for each region. To quantify the trends and associated uncertainties, a weighted least square approach was used (see Appendix [A.2.1](#) for details). The temporal trends of carbon in roundwood in all ecoregions were negative and greater than the residual variances. The average decreasing trend of all these regions was about 1.5%/yr. These findings were strengthened when the estimates of production for all years in this study were included.

To corroborate this finding, the trends of other factors that are known to be closely related to roundwood production were examined, such as the number of roundwood processing mills and the historical forest landownership pattern. From 1990 to 2007, the number of roundwood processing mills in NC decreased at an annual rate of 4.7 %/year ([North Carolina Division of Forest Resources, 2010](#), chap. 4). Result of the decreasing trend of roundwood production shown in this study is also consistent with the change in the timberland ownership distribution. In fact, the timberland ownership in NC has shifted: while the percentage of the forests owned by the private individuals did not change much, the industrial holding of the forest land has decreased by 7% between 2002 and 2007, which is part of the decreasing trend since 1980. Moreover, the share of public landownership has increased by about 10% during the same period ([Brown & New, 2012](#)). The management goal of industrial forestland owners is typically to apply intensive forest management to harvest timber, while that of the public owners is to provide collective benefits, such as those from biodiversity and ecosystem services ([Brown & New, 2012](#)).

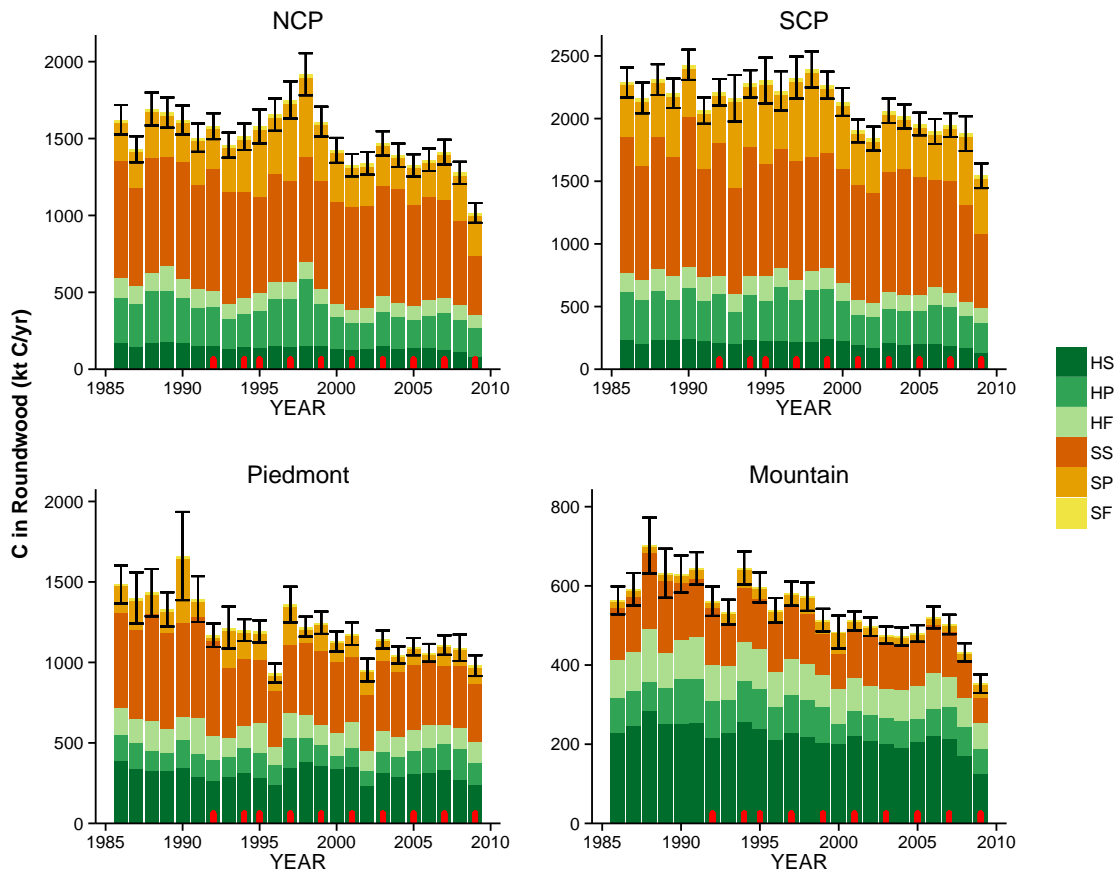


Figure 2.2: Time series of carbon in roundwood, with contribution from different types from 1986 to 2009 by ecoregion. Red tick marks on the x-axis indicate the years when the TPO data are available. Error bars represent 95% confidence interval for the total estimated carbon content. Average percentage change per year of carbon influx to HWP, \pm margin of error, for each ecoregion were as follows: NCP: -1.09 ± 0.64 %/year; SCP: -1.06 ± 0.43 %/year; Piedmont: -1.77 ± 0.60 %/year; Mountain: -1.79 ± 0.50 %/year. The Forest Service divides North Carolina into four regions: Northern Coastal Plain (NCP), Southern Coastal Plain (SCP), Piedmont, and Mountain. Types of roundwood are: HS=Hardwood Sawlog; HP=Hardwood Pulpwood; HF=Hardwood Fuelwood; SS=Softwood Sawlog; SP=Softwood Pulpwood; SF=Softwood Fuelwood. Plots are generated using the *ggplot2* (Wickham, 2009) package in R.

2.3.4.4 Comparison with other estimation methods

The comparison of the annual time series of carbon in roundwood in different ecoregions estimated using the proposed approach with that obtained using the *conversion factor* and the *linear interpolation* of the TPO record is shown in figure 2.3 (see section 2.3.3 for the rationale).

The results show that the estimates using the *conversion factor* approach range from values of about 1/3 of that using either the *linear interpolation* approach or the proposed estimation method for the Mountain region to about 1/12 of that in the Northern Coastal Plain (NCP) region (figure 2.3). Between 1992 and 2009, the carbon in roundwood estimated by *linear interpolation* in each ecoregion is always higher than that estimated in this study by 0.33 Mt C/yr or 6.2%/yr. The comparison based on ecoregions is shown in Appendix table A.5.

One of the disadvantages of the linear interpolation method is that it cannot estimate beyond the available years of the forest inventory data. When the inventory data is limited, the extension to the linear interpolation method is to combine with extrapolation. Adams et al. (2006) used both linear interpolation and extrapolation to estimate the annual roundwood production for different regions in the US. Their method was applied to NC when there are only three years of TPO data available and compared their extrapolation results with the estimates in this study (see section Appendix A.5.1 for details). The results (see tables A.3 and A.4) show that the estimation results were substantially better than the estimation using the linear

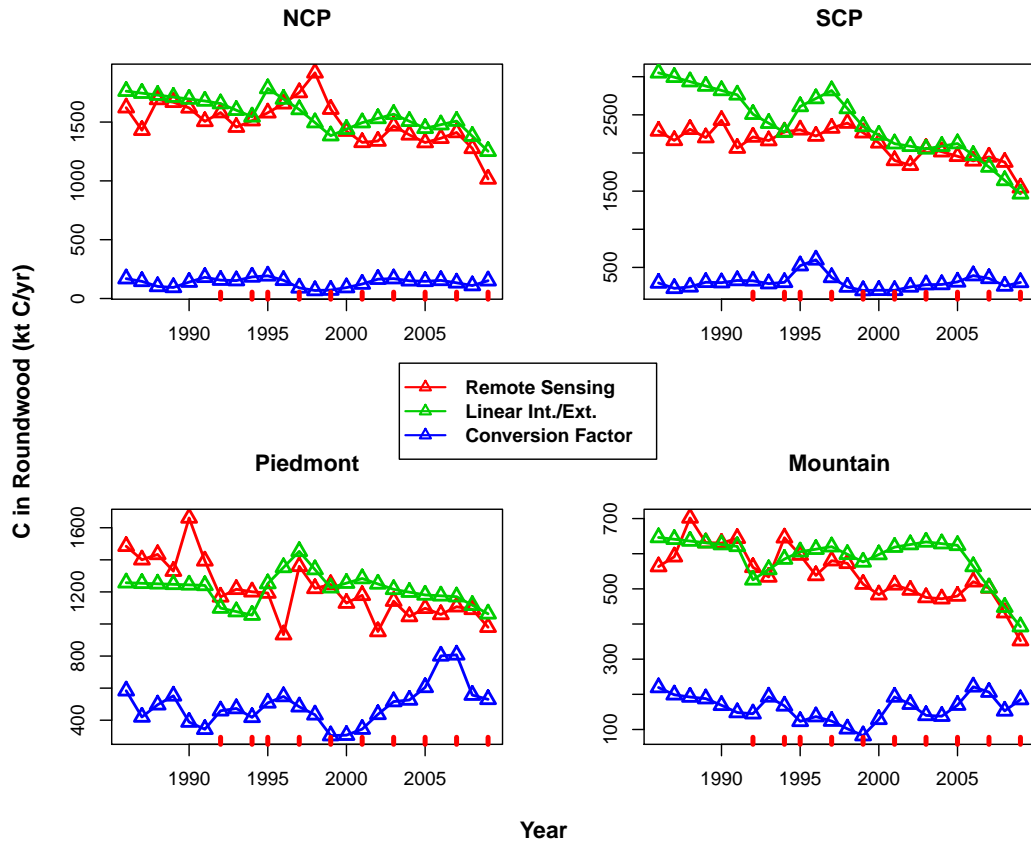


Figure 2.3: Time series of carbon in roundwood for different ecoregions using different estimation approaches. The “Remote Sensing” legend label refers to the estimation method detailed in this paper; The “Linear Int./Ext.” label refers to linear interpolation and extrapolation used to fill gaps in the TPO survey data; The “Conversion Factor” label refers to the estimation method based on the conversion factor conversion factors between areas affected by harvest and roundwood production volumes as calculated by Masek et al. (2011) to estimate roundwood production based on forest disturbance area. Red tick marks on the x-axis indicate the years when the TPO data are available. The Forest Service divides North Carolina into four regions: Northern Coastal Plain (NCP), Southern Coastal Plain (SCP), Piedmont, and Mountain.

interpolation–extrapolation method when only a few years of the TPO survey data are available for calibration. In addition, the estimates of roundwood production in some counties were negative when extrapolation was used. When linear extrapolation of the TPO data as taken into account, from 1986 to 2009, the estimate was 0.40 Mt C/yr or 7.4%/yr higher than the estimate in this study.

2.4 Uncertainties and limitations

To account for the uncertainty associated with the proposed model, the 95% confidence interval for each of the county-level estimates was calculated and propagated these uncertainties into the ecoregion-level analysis. In general, the uncertainty in the estimates will reflect the quality of the data and the amount of information available in terms of observations and variables (see Appendix [A.2](#) for details).

Another potentially significant source of uncertainty in the model is represented by natural disturbances. The regression approach used in this study, linking timber production to different types of disturbance intensity index using nonlinear interaction terms and area specific circumstances is basically used to calibrate the model. Because of this built-in flexibility, the proposed framework can adapt to different circumstances. For the case study in this chapter, besides harvest, the most common forest disturbance is related to wind storms ([Masek et al., 2011](#); [Williams et al., 2014](#)). Natural disturbance events in North Carolina, typically hurricanes, was tested whether they would affect the estimates. Results show no evidence that

hurricanes adversely affect the predictive power of the model (see Appendix [A.6](#) for details). In the proposed framework, if more detailed and specific information on natural disturbance events is available, the prediction accuracy can improve substantially. For example, if to apply to regions with high fire occurrence, such as California, additional data, such as the fire mask from the LANDFIRE program will be needed to mask out the areas disturbed by fires.

The data requirements limit the application of this model to areas where there are more than two years of subnational level, roundwood production survey data for the different wood types. Also this model may not provide accurate carbon estimates in places where land tenure is not well defined or where significant unreported logging occurs.

2.5 Discussion and conclusions

2.5.1 Implications for forest carbon accounting

This study demonstrates the use of remote sensing information and local wood production survey data, coupled with statistical model to estimate of annual carbon influx from forest management activities to the HWP pool. It translates disturbed forest areas at fine spatial resolution into carbon influx to the HWP pool, which is critical for assessing the impact of forest disturbance on the carbon cycle as it can detect management activities happened in smaller areas ([IPCC, 2014](#), p. 2.58). The proposed approach can also help in addressing one of the important goals of

the North American Carbon Program and the CarbonNA project (Kasischke et al., 2013) that has so far proven to be a major challenge in regional and global carbon cycle research as pointed out by (Liu et al., 2011). The model produced results that are comparable with other down-scaled models (for comparison) estimation results such as those by Zhou et al. (2013) and also with other indicators, such as decreasing in the number of wood processing mills and timberland areas. Other conventional estimation methods, such as those based on conversion factors for forest area changes to carbon in roundwood or linear interpolation of missing data, either do not provide sufficient spatially detailed estimates or are not reliable when only limited wood production survey data is available.

Both *linear interpolation* and *conversion factor* have substantial shortcomings. Linear interpolation does not have a sound physical basis for estimating production in missing years, as it does not provide any link between forest disturbance events and roundwood production, and cannot be used outside the range of years used for calibration. The conversion factor method is based on coarse-scale average roundwood harvest rate in volume per hectare and does not distinguish between the production of different roundwood types. One application that more general methods cannot address, is the 30m Landsat resolution which, as suggested by the IPCC Guidelines (IPCC, 2014, section 2.2.6.2), is able to link large area processes such as regional carbon sequestration with the human activities smaller than 1 ha that are missed in all other methods. In this study the carbon in roundwood produced was estimated. Since not all of the roundwood produced ends up as

primary wood product, due to wood residues left in the mill during processing, this is an upper limit of the forest carbon influx to the HWP in-use or in landfill. The estimate, when combined with data from the lookup tables as from the USDA report, that return the average amount of carbon in HWP that ends in use/landfill over time after harvest (Hoover et al., 2014), can provide a link between forest management activity and carbon stored in HWP.

2.5.2 Policy implications

Estimation method proposed in this chapter can help countries to fulfill the newest IPCC estimation guideline to include the contribution of HWP to the change of carbon stock in forests, because it provides a link between the Carbon in HWP and forest management activities (IPCC, 2014, section 2.8.1.2). The guidelines were developed to support the implementation of the second commitment period of the Kyoto Protocol (Decision 2/CMP.7)⁸ which recommends to “explore more comprehensive accounting of anthropogenic emissions by sources and removals by sinks from land use, land-use change and forestry” for better forest carbon accounting. In the proposed framework framework, countries can estimate HWP production coming from domestic forest-related activities (step 2.1, section 2.8.1 in the IPCC Guidelines (IPCC, 2014)). Once the deforested areas are identified, besides using the ratio of the deforested area to the total disturbed area (equation 2.8.3 in the IPCC

⁸The document is available at <http://unfccc.int/resource/docs/2011/cmp7/eng/10a01.pdf>

Guidelines(IPCC, 2014)) to derive the annual fraction of HWP carbon contribution from deforestation⁹, the country can also use the proposed approach to achieve this by evaluating the estimated regression function using the deforested areas only as input. If roundwood harvested as a result of deforestation is also reported in the forest inventory data, then the estimate could be more accurate.

The proposed modeling framework could help countries to develop domestic management policies aimed at meeting their carbon storage goals in both standing forest biomass and HWP. For example, [Zhu et al. \(2010\)](#) and [Lippke et al. \(2011\)](#) envision a multidisciplinary approach that includes remote sensing in assessing carbon dynamics (including HWP carbon) in land-use change and land-management activities. The approach could also encourage the adoption of Kyoto type carbon accounting frameworks as the inclusion of HWP could help local forest-based industries in timber-rich countries. In this sense, this improved modeling framework can be used to provide new information to contribute to a more complete complete carbon crediting and trading system as the one proposed by [Ellison et al. \(2011\)](#).

⁹According to the 2013 IPCC Guideline, “Deforestation” is the direct human-induced conversion of forested to non-forested land (section 2.6.1).

Chapter 3: Spatially Explicit Assessment of the Impact of Forest Management Regimes on Forest Ecosystem Services in a Species-Rich Area

Abstract To quantify the impact of different management regimes on selected ecosystem services, specifically, carbon sequestration, roundwood harvested, and forest habitat provision for sustaining biodiversity in a species-rich area, a spatially-explicit, mechanistic model was applied at the Grandfather Ranger District, located in the Southern Appalachians in North Carolina, USA. The management regimes consist of 2 baseline scenarios: no harvest with complete fire suppression, and no harvest with no fire suppression; 2 harvest methods: various level of clearcutting (Aggressive) and various level of selective harvest (Moderate). The 2 harvest methods are simulated for scenarios with both no fire suppression and complete fire suppression. The results show that i. although the Aggressive regimes yield about 15 times more harvested roundwood, such regimes result in the forest average tree age and standard deviation of tree age to be about 20 years younger than those in the baseline scenarios and the Moderate regimes; ii. the Aggressive regimes also result in a higher proportion of both ecologically desirable and invasive species; iii.

Active planting plays a more important role in increasing the proportion of ecologically desirable species than opening up the canopy as a result of fire. iv. When fire is not suppressed, the forest aboveground biomass becomes a carbon source; v. The Moderate regimes have higher annual carbon sequestration rate in the aboveground forest biomass for about 12% when fire is completely suppressed. vi. Certain ecozones are especially sensitive to fire and the Aggressive regimes. This case study shows that a spatially explicit, mechanistic model applied is especially helpful in quantifying ecosystem service trade-offs in a species rich forest. It not only allows multiple management regimes applied to various locations in a single scenario, but also provides quantitative spatial information on the relative number of species present and age composition. More comprehensive understanding of the impact of different management regimes on the quality of the forest habitat was gained, which allows forest managers to design some innovative management regimes and policy makers implement some form of crediting system that can balance the ecosystem goods and services that a forest provides. Additionally, the ability to distinguish between sawlog and pulpwood harvested allows more accurate estimate on the income of the forest owners and better account for forest carbon cycle.

3.1 Introduction

To effectively manage a forest such that the goods and services provided are balanced, quantitative understanding the impact of different forest management regimes on forest dynamics is essential. Particularly the interaction and feedback between the ecological processes and the management regimes applied, because it can result in spatial and temporal difference in the trade-offs (Seppelt et al., 2011). Having a robust understanding of such feedback and interaction can help predict if a particular policy instrument would succeed or fail (Carpenter et al., 2009). The forest ecosystem services focused in this study are carbon sequestration, roundwood harvested, and provision of habitat to sustain biodiversity.

The relative effect of specific management schemes, such as the choice of species and method of harvesting on ecosystem carbon sequestration over time remains a major uncertainty in measurement of the terrestrial carbon sink (Davis et al., 2009; McKinley et al., 2011; Nunery & Keeton, 2010), especially in temperate forests. In addition to landowners management practices, natural disturbances such as wind storms and fire also have important impact in changing forest carbon dynamics and vegetation structure (Running, 2008). A systematic way of understanding the linkage between forest management decisions, natural disturbances, and the carbon cycle can help further analysis of the possible fate of the forest carbon cycle with different management practices.

Some management schemes favor one aspect of ecosystem service over the

others ([Schwenk et al., 2012](#)). For example, intensive management like clear-cutting, would yield the highest volume of roundwood harvested, but results in a large patch of even-aged trees, which could negatively impact the habitat quality ([Keenan & Kimmins, 1993](#)). A management regime that focuses on carbon sequestration may be at the expense of other ecosystem services, such as timber production and habitat quality ([Schwenk et al., 2012](#); [Seidl et al., 2007](#)).

Results from model simulations can inform policy makers of the impact of different management options on the forest. Information that would be the most useful to forest managers must be spatially explicit and at the landscape scale [Müller et al. \(2010\)](#); [Scheller & Mladenoff \(2007\)](#), because only these allow managers to understand the effect of the treatments in different areas, especially areas with special conservation interests ([Gustafson, 2013](#); [Scheller & Mladenoff, 2007](#)). The two major advantages of using a mechanistic model in comparison to an empirical model are 1. different conditions, such as management regimes and climate pattern, that have not happened before can be simulated; 2. the interaction between the management or disturbances and the underlying physical processes of forest succession can be better understood. Those are important elements for analyzing the impact of the new management regimes on ecosystem services trade-offs, the location of the trade-offs, and the possible interaction and feedback between human modification and the ecosystem ([Howe et al., 2014](#)).

Two harvest regimes, along with the decision of fire suppression, were simulated to understand the possible ecosystem service trade-offs. The working hypothe-

sis is that clearcutting is the more advantageous for profit-oriented forestry, because it is more cost effective than selective harvest ([Keenan & Kimmins, 1993](#)). An alternative harvest regime that selective harvests slightly less area than the clearcut and aims at both preserving the environment and sustainable roundwood harvest was tested whether it would benefit the habitat and carbon sequestration at the expense of the amount of roundwood harvest.

The objectives of this study were to understand the management impact on 1. carbon sequestration by the standing forest biomass; 2. the amount of roundwood (sawlog and pulpwood) harvested over the 100 years for each management regime; 3. forest species composition and age structure. The working hypothesis is that an improved forest management scheme, including harvesting and the choice of species planted, could achieve higher level of carbon sequestration ([Pan et al., 2011](#); [Ruddell et al., 2007](#)) and habitat quality ([Kusumoto et al., 2015](#)) compared to business-as-usual.

3.2 Material and methods

3.2.1 Study area

To understand the impact of different management regimes coupled with species competition, a region that is located in a species-rich area that has an active timber industry and facing increasing pressure to harvest commodities was chosen([Fox et al., 2007](#)). The Grandfather Ranger District (GRD) within the Pisgah

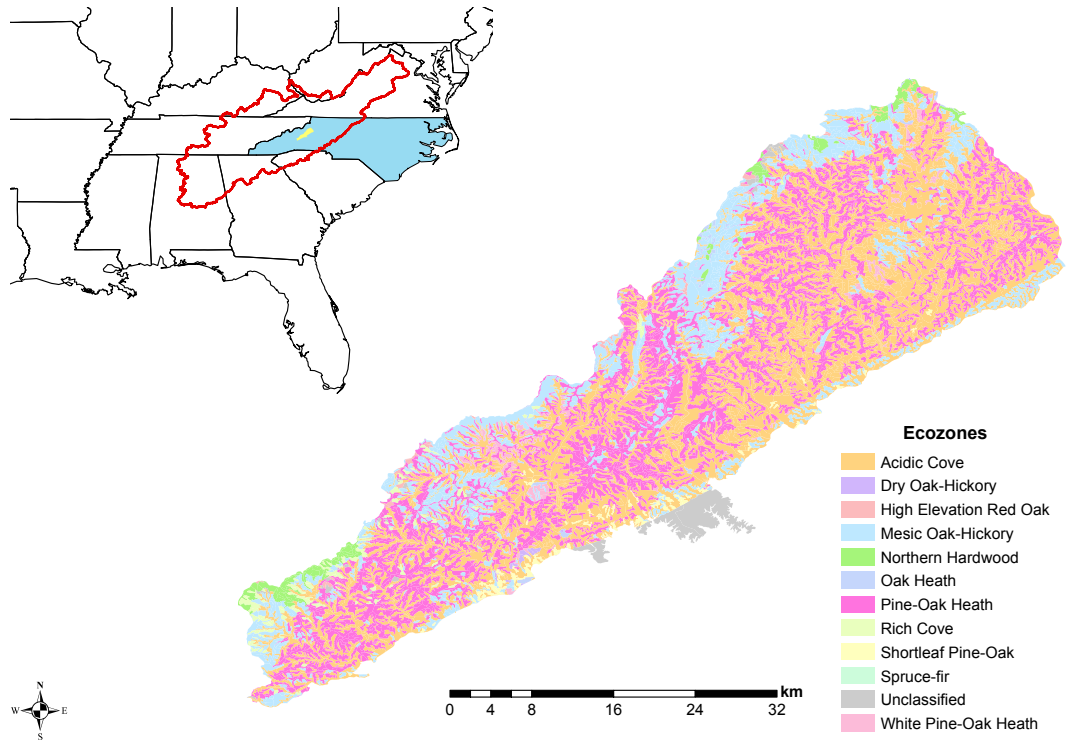


Figure 3.1: Ecozones in the Grandfather District (highlighted in yellow), North Carolina, USA. Red boundary indicates the Southern Appalachian Region.

National Forest is about 777 km^2 and is located in the eastern edge of the Southern Blue Ridge Ecological Province, in North Carolina, southeastern USA (Fig. 3.1).

The elevation of the GRD ranges from 314m to 1810m and it changes sharply within a short distance: 1200m vertical change in less than 6.5 km horizontal distance (U.S. Department of Agriculture, Forest Service National Forests in North Carolina, 2011). The sharp difference in topography results in diverse vegetation composition (Pittillo et al., 1998) that supports high richness of animal diversity. The GRD consists of 12 ecozones (Simon et al., 2005) (Fig. 3.1), each has its own ecological concern. In particular, ecozones “Pine-Oak Heath”, “White Pine-Oak Heath”, “Shortleaf Pine-Oak”, “Rich Cove” and “Acidic Cove” all depart from a

desirable condition to a very high degree: either with a large percent of invasive species, undesirable species composition, or with tree stands that are even-aged due to historical management methods ([U.S. Department of Agriculture, Forest Service National Forests in North Carolina, 2011](#)).

Field data

The field data used in this study were obtained from the US Forest Service (USFS) Forest Inventory Analysis (FIA) data version 6.0.1 ([U.S. Department of Agriculture, Forest Service, 2016](#)), available for 1974 to 2015. The FIA data consists of assessment of field plots, each 1 acre (0.4 ha), that provide information on the entire plot, comprising tree age, species, site condition, forest type, and estimated biomass for individual trees. An acre (0.4 ha) sample is assessed, on average, for 6000 acre (2428.1 ha) of forest and was designed to provide an unbiased representation of forest types. Plots for the eastern states are censused every 5-7 years. This study only uses the plots that are located in forestland, from 2002 to 2015.

Precise locations of the plot centers are not provided: owing to the need to avoid identification of specific landowners and to protect the plots from damage. Locations are provided within 1 mile (1.6 km) of the exact plot location. Some plots located in the private forest are swapped with other plots of similar conditions within the same county. Since the plots are swapped with nearby plots of similar forest conditions, the data remain valid for the forest type measured. Therefore, not having the precise location information does not affect the use of the FIA data in

this study. Biomass is estimated using the allometric equation provided by [Jenkins et al. \(2003\)](#).

Species

Twenty of the most abundant tree species, according to the FIA data provided in the southern Appalachian region in North Carolina were chosen in this analysis (table 3.1). Ecological and commercial ranks were provided by local experts, based on the species contribution to its community and on the amount of volume sold and products made respectively. Based on the ranks, species were classified into 4 ecological classes, in which Class A is the most ecologically preferable, while Class D is the least. Both Class A and Class D contain 3 species each. Class A consists of Pitch Pine, White Oak, and Eastern Hemlock. Particularly, Eastern Hemlock is a keystone species in riparian forests and acidic coves. Species are also classified into 5 commercial classes, in which Class I is the most economically valuable, while Class V is the least. Specifically, the three species in Class D: Yellow Poplar, Eastern White Pine, and Red Maple are aggressive native species with capability to displace other species, in which, the first two are also the top 2 ranked commercial species.

Table 3.1: The 20 species simulated in this study. “Comm Rank” and “Eco Rank” stand for “Commercial Rank” and “Ecological Rank” respectively. They are provided by local experts based on resulting harvested timber products and the contribution to its community of each species. Smaller rank index indicates higher rank. “Comm Class” and “Eco Class” group the commercial rank and the ecological rank into 5 and 4 classes respectively.

Common Name	Scientific Name	Code	Comm Rank	Comm Class	Eco Rank	Eco Class
Pitch Pine	<i>Pinus rigida</i> Mill.	PIRI	9	III	1	A
White Oak	<i>Quercus alba</i> L.	QUAL	3	I	2	A
Eastern Hemlock	<i>Tsuga canadensis</i> (L.) Carr.	TSCA	16	III	3	A
Sweet Birch	<i>Betula Lenta</i> L.	BELE	8	II	4	B
Table Mountain Pine	<i>Pinus pungens</i> Lamb.	PIPU	11	III	5	B
Chestnut Oak	<i>Quercus prinus</i> L.	QUPR	6	II	6	B
Northern Red Oak	<i>Quercus rubra</i> L.	QURU	4	I	7	B
American Beech	<i>Fagus grandifolia</i> Ehrh.	FAGR	13	IV	8	B
Pignut Hickory	<i>Carya glabra</i> (Mill.) Sweet	CAGL	14	IV	9	B
Sourwood	<i>Oxydendrum arboreum</i> (L.) DC.	OXAR	19	V	10	C
Flowering Dogwood	<i>Cornus florida</i> L.	COFL	20	V	11	C
Blackgum	<i>Nyssa sylvatica</i> Marsh.	NYSY	15	IV	12	C
Black Cherry	<i>Prunus serotina</i> Ehrh.	PRSE	7	II	13	C
Black Locust	<i>Robinia pseudoacacia</i> L.	ROPS	18	V	14	C
Scarlet Oak	<i>Quercus coccinea</i> Muenchh.	QUCO	17	IV	15	C
Black Oak	<i>Quercus velutina</i> Lam.	QUVE	5	I	16	C
Virginia Pine	<i>Pinus virginiana</i> Mill.	PIVI	10	III	17	C
tuliptree/ yellow poplar	<i>Liriodendron tulipifera</i> L.	LITU	1	I	18	D
red maple	<i>Acer rubrum</i> L.	ACRU	12	IV	19	D
eastern white pine	<i>Pinus strobus</i> L.	PIST	2	I	20	D

3.2.2 Management scenarios and targets

Six management scenarios were simulated, which consists of 2 baseline scenarios when no harvest occurs—no disturbance and fire is the only disturbance, 2 scenarios with different harvest regimes when fire is complete suppressed or completely allowed (table 3.2). The 2 harvest regimes are “Aggressive” and “Moderate”. The “Aggressive” regime consists of clear-cutting and shelterwood harvesting. Clearcut occurs in lands that are privately owned and in publicly owned non-ecologically sensitive lands; shelterwood harvest occurs in lands that are ecologically sensitive and publicly owned lands. The “Moderate” regime mimics the practice of the US Forest Services in the National Forest for the public forest, which include shelterwood harvest and selectively harvest certain species in different areas. Managers in privately-owned forest would conduct selective logging for the 3 economically valuable tree species and replant them, but the area of harvest varies depending on its protection status classified by the North Carolina GAP study (McKerrow et al., 2006). No harvest takes place in the Congressional Designated Roadless areas, where the federal law prohibits any forest management activity such as logging or road construction to occur. Harvest occurs every 5 years for both of the harvest regimes. Details of the harvest regimes used in each specific management area are provided in Appendix table B.1. This study adopts the current harvest rotation of the Southern Appalachian area of 80 years (Fox et al., 2007), which means that all harvested sites grow at least 80 years since its previous harvest.

Table 3.2: Two harvest regimes simulated in this study.

Regimes	Private	Public	Ecology	Roadless
Aggressive	Clearcut 2.5% area per year: Select most economically valuable sites + Plant the 3 most economically valuable species	Clearcut 1-2.5% area per year: Select most economically valuable sites + Plant the 3 most ecologically preferable species	Shelterwood harvest 0.5% area per year: Select most economically valuable sites + Plant the 3 most ecologically preferable species	No harvest
Moderate	Selectively cut 0.1-2.5% area per year: Select most economically valuable sites + Plant the 3 most economically valuable species	Shelterwood harvest + Planting Red Oak and Pitch Pine or 1-2% area per year: Selectively harvest the 3 most ecologically undesirable species of 0.4 ha for each site	0.5% area per year: Cut understories or + 0.24% area per year Shelterwood harvest and Planting Red Oak and Pitch Pine	No harvest

Fire was the most frequent natural disturbance in the Southern Appalachian region before the USFS started suppressing fires. The complete fire suppression regime that the USFS currently implements (Flatley et al., 2013) was simulated, and, for comparison, also the natural fire regime based on the historical fire data in Xi et al. (2009). All management scenarios were simulated with and without fire suppression. Also, scenarios of complete fire suppression (no disturbance) and no harvest (no fire suppression) were simulated to assess the cumulative impact.

3.2.3 Model description

Forest change over 100 years was simulated using a spatially explicit, forest succession model, Landis-II, for each of the management scenarios described above (table 3.2). The 150m resolution was chosen to balance between the need of including the interactions between the ecological processes and the forest management regimes applied and computing limitations. Landis-II is a mechanistic forest succession model that can be used to simulate forest response to different disturbances and provide spatially explicit results that are based on the interactions of species productivity, mortality, reproduction, natural disturbances and management activities (Scheller et al., 2007; Scheller & Mladenoff, 2004). Since it is a spatially explicit model, Landis-II can simulate inter- and intra-species competition at a broad scale ($> 10^5$ ha), including processes such as seed dispersal in the landscape and other succession processes (Scheller et al., 2007). To account for the carbon and biomass dynamics in the forest during the succession process, the Forest Carbon Succes-

sion extension version 2.0 (Dymond et al., 2016) was used. Detail studies of the sensitivities of the model are given by Simons-Legaard et al. (2015) and de Bruijn et al. (2014) and in particular they identified the maximum biomass and maximum ANPP, were the parameters to which the model was most sensitive.

The two extensions to the core model were used to simulate the impact of the harvest and fire disturbances were the Base Harvest and the Base Fire. The Base Harvest version 3.0 allows the simulation of different harvest methods, areas harvested, and rotation length (Gustafson et al., 2000). Wildfires are generated based on the physical characteristics of each ecozone in the Base Fire extension version 3.0.3 (He & Mladenoff, 1999). Details on how the model input and parameters were obtained are provided in Appendix B.

3.2.4 Impact on carbon sequestration, roundwood harvested , and habitat provision

The impacts of different management schemes on the 3 ecosystem services were quantified using the output from the model to understand the trade-offs. Carbon sequestration in the aboveground standing biomass is defined by the net change of carbon in the forest biomass in between simulation years 1 and 100. The net change was divided by 100 to obtain the annual carbon sequestration by the aboveground forest biomass.

The Forest Carbon Succession Extension in Landis-II provides information on

the carbon in the trunk and branches of the tree that goes to the harvested product pool (Dymond et al., 2016). The volume of roundwood harvested is estimated by converting the annual carbon flux to the harvested product pool of each species to volume. The roundwood specific gravity used to convert from biomass to volume is the average specific gravity of both hardwood and pulpwood in the Southeastern US, $0.49 \text{ m}^3/\text{g}$ (Smith et al., 2006). Harvested roundwood is divided into sawlog and pulpwood harvested. For species that belong to commercial classes I-III (table 3.1), the harvested trunk was considered as harvested sawlog while the harvested branches portion was considered as harvested pulpwood. All the harvest for Virginia Pine and for species that belongs to commercial classes IV and V is considered as harvested pulpwood.

Provision of habitat to maintain biodiversity is more difficult to quantify. The Shannon's diversity index (Shannon, 1948) was used to assess the tree species abundance and richness after 100 years. However, the Shannon's diversity index alone does not account for the ecological values of species (Hurlbert, 1971) and the species age structure, both are important indicators of the desirability of a forest habitat that can sustain biodiversity (Noss, 1999; Tews et al., 2004). Therefore, in addition to reporting the Shannon's diversity index, the frequency distribution of all species for the entire GRD, the average and the standard deviation of age of all species for the entire GRD and individual ecozones at the end of the simulation year for all management scenarios were analyzed. The preferred condition is a high number of Class A species, low number of Class D species, older age, and high standard

deviation of age that indicates a high age diversity.

3.2.5 Calibration, validation, and sensitivity analysis

The predictive ability of the Landis-II model depends on how close the simulated forest succession results were to field data. Calibration was done at the plot level and also at the forest type level for each species, while validation was done at the plot level. For the GRD, there are total of 9 forest types. For 2002 to 2015, each forest type has about 4 to 5 plot-level aboveground biomass values, for a total 50 data points. At the species-level aboveground biomass, each species is present in at least one forest type for one year for FIA data ranges from 2002 to 2010. On average each species is present in 5 forest types for 3 to 4 years, with total of 367 data points. The Forest Carbon Succession Extension of the model was simulated for 14 years. The average biomass of the first 9 years of the simulation result was used to calibrate with that from the FIA data version 6.0.1 ([U.S. Department of Agriculture, Forest Service, 2016](#)) in between 2002 and 2010. The simulation results of the later 5 years were used as validation by comparing against the values of the 2011 to 2015 FIA data. Data used for both calibration and validation are located in areas that had not experienced any forest disturbance between 1990 and 2010. Areas that have not had any disturbance event occurred between 1990 and 2010 were selected for calibration of the Forest Carbon Succession extension. Maps of the disturbance history in the GRD were obtained from the forest disturbance history maps derived from the Vegetation Change Tracker (VCT) algorithm ([Huang](#)

et al., 2010). The resulting biomass time series from the simulation of Forest Carbon Succession extension was converted to dry biomass using the species-specific Specific Gravity values in Miles & Smith (2009).

For sensitivity analysis, the maximum ANPP and the maximum biomass of each species were altered by 10% in the calibration process, as those two parameters are shown to be the most influential in the model (Simons-Legaard et al., 2015; Thompson et al., 2011). The changes in the aboveground biomass with the original data in simulation years 0, 50, and 100 were compared. The initial data for maximum biomass of each species were obtained based on the data used in Thompson et al. (2011). Values for maximum ANPP were obtained from simulation of the PnET-II model de Bruijn et al. (2014); Xu et al. (2009). Details for input parameters used in the PnET-II model are available in the Appendix B.13.

3.2.5.1 Model uncertainty and statistical analyses

Each management scenario is simulated for five times to account for the between-run variability, which captures the stochastic component of Landis-II (Thompson et al., 2011). The range of the model output values can be used to compare the significance of the difference between management regimes. The contribution of the uncertainties in input parameters were analyzed using sensitivity analysis. A more comprehensive sensitivity analysis for all the input parameters was done by Simons-Legaard et al. (2015), albeit in a null landscape in North America.

The Shannon's diversity index, used as a metric of diversity, is on a logarithmic

scale that indicates the entropy of the area rather than other aspects of diversity, so it is difficult to compare the values between management regimes. To compare the magnitudes of the Shannon's diversity index of two scenarios, the effective numbers of species, as calculated by taking the exponent of the Shannon's diversity index, of two scenarios were calculated to compare their differences on a linear scale (Jost, 2006; MacArthur, 1965). The student t-test was used to compare the significance of the difference between the index in any two scenarios (Hutcheson, 1970).

3.3 Results

3.3.1 Calibration, validation, and sensitivity analysis results

For the plot-level calibration and validation, the distributions of the plot-level aboveground biomass density of the FIA data and the model simulation were compared (Fig. 3.2) and the student t-test was used to compare the means of the two distributions. For the FIA data used for calibration, the mean was 140.2 t/ha and that of the model simulation is 141.0 t/ha. The 95% confidence interval was -17.8 and 16.2, and t-value was -0.09. Since the t-value lies within the confidence interval and the p-value was 0.93, the t-test results show that the hypothesis that the means of the two distributions are equal cannot be rejected.

In addition to the aboveground biomass density at the plot-level, that at the forest-type level per species was also examined to ensure the species-level aboveground biomass density is calibrated. The plot of the simulation aboveground

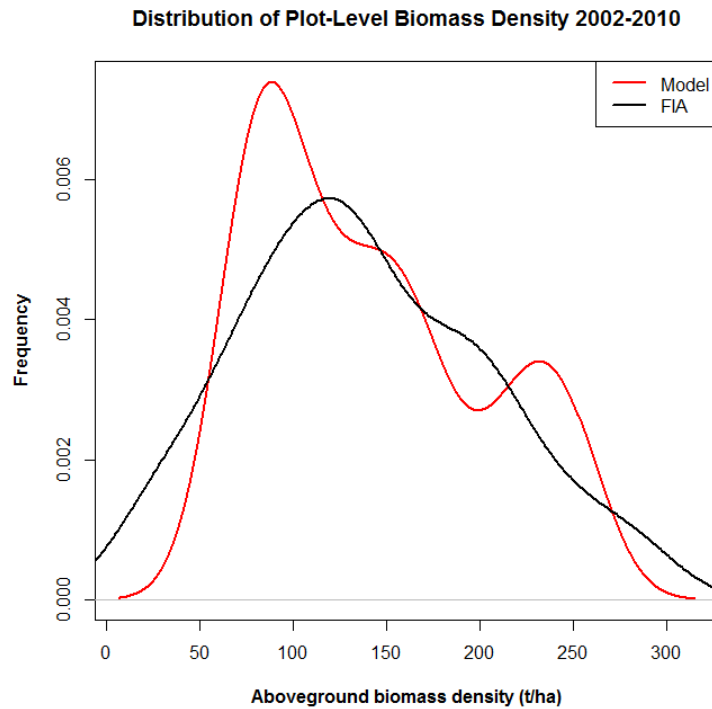


Figure 3.2: Model was calibrated using the plot-level aboveground biomass of the FIA data for 2002-2010 (black line, mean=140.2 t/ha) and simulation results for the first 9 years (red line, mean=141.0 t/ha). The two lines show the Kernel density estimation of the 2 distributions. The results from the Student t-test indicate that the two means are not significantly different.

biomass density of the first 14 years against the corresponding FIA data (Fig. 3.3) shows that the relationship between the simulated result and the FIA data is generally good: Slope of the regression line is 0.90, and 95% confidence interval of the slope was 0.78 and 1.03, where 1 lies within the interval. The adjusted r-square of the regression line is 0.65, with p-value less than 0.01. Although there are some underestimations of Northern Red Oak (QURU) and Yellow Poplar (LITU) for some area and overestimation Sweet birch (BELE) for some data point, most of the data points fall within the 95% prediction interval. Note that about half of the FIA data points are distributed around the lower end of the aboveground biomass density. Both of the species-level aboveground biomass density of the FIA data and the simulated model results are not normally distributed (Shapiro-Wilk test was performed on both data set, with both of the p-values less than 0.01, which implied the null hypothesis that the data sets are normally distributed are rejected). However, the number of data points for both datasets is 109. With that few number of data point and the GRF is a small region in the Southern Appalachian, it is not unreasonable to have datasets that are not normally distributed. Nevertheless, since all studied species and forest types are included in the calibration, the FIA data used in the calibration have provided sufficient information.

For validation, the simulated average aboveground biomass density value is 162.5 t/ha while that of the FIA is 171.7 t/ha (Fig. 3.4). The t statistics is -0.57, which lies within the 95% confidence interval of -41.5 and 23.0, and p-value was 0.57. The t statistics results show that the hypothesis that the means of the two distri-

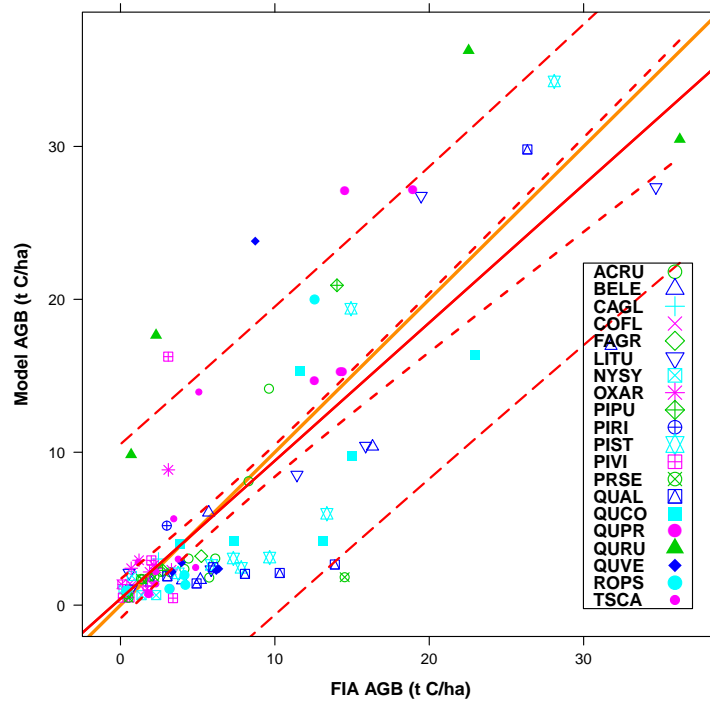


Figure 3.3: Comparison of the modeled aboveground biomass and the FIA aboveground biomass for 13 years of simulation for places where no disturbance had occurred in between 1990 and 2010. Total number of data points is 113. Each data point represents the average biomass of a species in a community in a given year. Species codes in the legend are identified in Table 3.1. Orange line represents the 1:1 relationship and the red line represents the linear regression of the model and the FIA data. Slope of the regression line is 0.65, with p-value less than 0.01. The outer pair of the red dash lines is the 95% prediction interval, while the inner pair is the 95% confidence interval.

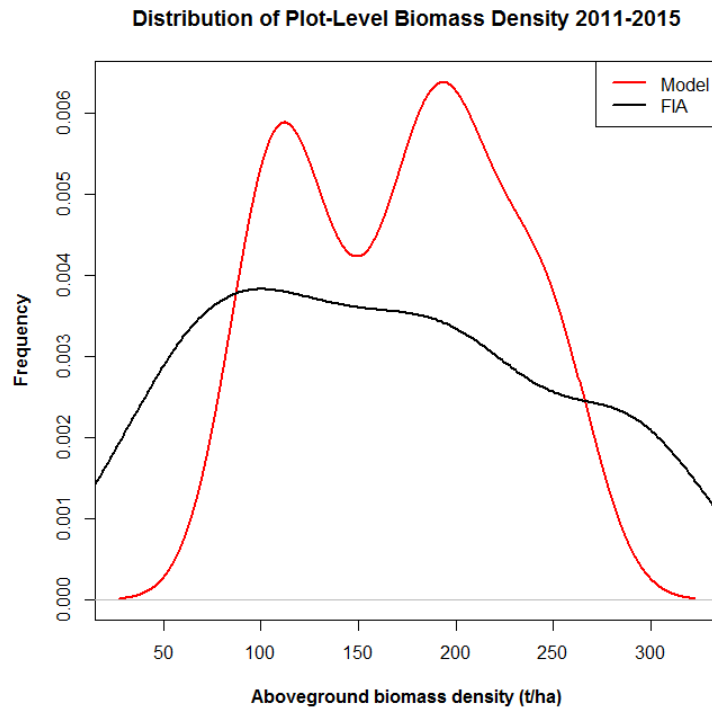


Figure 3.4: The plot-level aboveground biomass of the FIA data for 2011-2015 (black line, mean=162.5 t/ha) was used to validate the model results by comparing it with that for the last 5 years out of 14 years of the model simulation results (red line, mean=171.7 t/ha). The results from the Student t-test indicate that the two means are not significantly different. The two lines show the Kernel density estimation of the 2 distributions.

butions are equal cannot be rejected. Additionally, the aboveground forest biomass of the nearby undisturbed hardwood forest at the Coweeta Hydrologic Laboratory was 139.9 t/ha in the study conducted in the 1970s (Day & Monk, 1974). Although the study time period is different, the biomass of the model simulation for earlier years is 141.0 t/ha, which is similar to that at the Coweeta study site.

Maximum ANPP and maximum biomass were chosen for analyzing the sensitivity of the model. For 10% change in the maximum ANPP results in $\pm 0-2\%$ change in the aboveground biomass density, while 10% change in the maximum biomass re-

sults in $\pm 7.5\text{-}10\%$ change in the aboveground biomass density (Table 3.3). The changes of the influences over time of the two parameters are opposite from one another: While the change in the maximum ANPP value has less influence in year 50 when compared to year 0 and year 100, that for the maximum biomass has the most influence in year 50 when compared to the other 2 years.

Table 3.3: Sensitivity analysis of the change in aboveground biomass density (AGB) by altering two parameters for simulation years 0, 50, and 100. “Original”, the scenario which there is no forest disturbance (complete fire suppression) was served as the basis of comparison.

Parameter	Change	Year 1		Year 50		Year 100	
		AGB (t)	Change(%)	AGB (t)	Change(%)	AGB (t)	Change(%)
Original	0	222.24	0	319.15	0	369.35	0
Max ANPP	-10%	217.19	-2.27	318.18	-0.31	361.93	-2.01
Max ANPP	+10%	227.78	2.49	319.17	0.01	375.07	1.55
Max Biomass	-10%	205.52	-7.52	287.56	-9.90	338.34	-8.40
Max Biomass	+10%	239.40	7.72	349.86	9.62	398.76	7.96

3.3.2 Simulation Results

3.3.2.1 Effect of treatments on carbon sequestration

In this study, fire suppression is the most important factor in the management in terms of the impact on the carbon sequestration by the forest standing biomass after 100 years (Table 3.4). If fire is completely suppressed, all the management schemes result a net carbon gain in the standing biomass in the landscape. As expected, the Aggressive regimes, which consist of different level of clear-cutting or shelterwood harvest, make the forest be a smaller carbon sink or a bigger carbon source when compared with other management regimes or the baseline scenarios: when fire is suppressed, the aboveground biomass of the forest serves as an about 13%/yr smaller carbon sink in the Aggressive regime than those in the Moderate regime and the baseline; When fire is not suppressed, the forest release about 13%/yr more carbon in the Aggressive regime than that in the baseline scenario and about 8%/yr more compared with the Moderate regime. When fire is suppressed, the Moderate regime sequester similar amount of carbon compared with the baseline. But when fire is not suppressed, the Moderate regime releases about 5%/yr more carbon compared to the baseline. However, when sawlog harvested is considered, the loss of carbon in both harvest regimes when fire is not suppressed is indeed transferred to the long-term product pool (sawlog), where the carbon can be stored for about 100 years (Smith et al., 2006).

Table 3.4: Average annual carbon sequestration rate in the standing biomass in the 100-year simulated time period, average annual harvested sawlog, pulpwood densities, and harvested area over 100-year period for different management regimes. The Shannon's Diversity Index, the average and standard deviation of age of all the species are reported for the landscape at the end of the 100-year period. The reported values are the average \pm the standard deviation of the 5 model runs. Negative numbers indicate a loss.

Regime	Fire	Sequestered (t C/ha/yr)	Sawlog (m^3/ha/yr)	Pulpwood (m^3/ha/yr)	Harvested Area (ha/yr)	Diversity (Shannon's)	Avg. Age (yr)	SD Age (yr)
NoDist	N	0.74 \pm 0.00	0	0	0	2.32 \pm 0.00	73.13 \pm 0.02	45.23 \pm 0.01
Aggressive	N	0.64 \pm 0.01	3.62 \pm 0.02	2.60 \pm 0.02	522.65 \pm 14.77	2.47 \pm 0.00	57.52 \pm 0.70	43.23 \pm 0.12
Moderate	N	0.73 \pm 0.00	4.95 \pm 0.11	3.45 \pm 0.10	34.38 \pm 1.03	2.34 \pm 0.00	72.76 \pm 0.02	45.23 \pm 0.01
Fire	Y	-0.126 \pm 0.01	0	0	0	2.59 \pm 0.00	51.70 \pm 0.47	44.31 \pm 0.14
Aggressive	Y	-0.14 \pm 0.01	4.15 \pm 0.06	2.87 \pm 0.05	554.25 \pm 13.65	2.50 \pm 0.01	35.87 \pm 0.35	34.71 \pm 0.14
Moderate	Y	-0.13 \pm 0.02	4.73 \pm 0.19	2.69 \pm 0.16	34.80 \pm 1.48	2.61 \pm 0.00	51.20 \pm 0.47	44.09 \pm 0.33

The time of the aboveground biomass density (fig. 3.5) shows that fire is a major factor in determining the annual change of carbon in biomass. The trends of the biomass density of the Moderate regimes are very similar to the baseline scenarios. Although the Aggressive regimes have biomass density trends that are almost always lower than the counterparts, in around years 85-90, the biomass densities of the Aggressive regimes, whether fire is suppressed or not, are slightly higher than their counterparts. The reasons could be because in the Aggressive regimes, two out of the three most invasive plants are planted more frequently than in the Moderate regimes as they are among the most commercially desirable species. The minimum harvesting time for a stand is 80 years. By around years 85-90, the stands that were planted in the beginning of the model simulation would have had grown to mature, but were harvested shortly after, which can be observed especially in the time series of complete fire suppression.

3.3.2.2 Effect of treatments on roundwood harvested

Figure 2.1(b) from chapter 2 shows that the harvest density of the forestland in the corresponding counties of the GRD in between 1986 and 2010 is about $0.08m^3/ha/yr$ to $3.67m^3/ha/yr$, which is similar to those found in this chapter (table 3.4). Although the volume of roundwood harvested per hectare in both of the Moderate regimes harvested is more than that in the Aggressive regimes, when accounting for the actual areas that were harvested, the Aggressive regimes actually yield more roundwood volume (table 3.4). When fire is completely suppressed, the

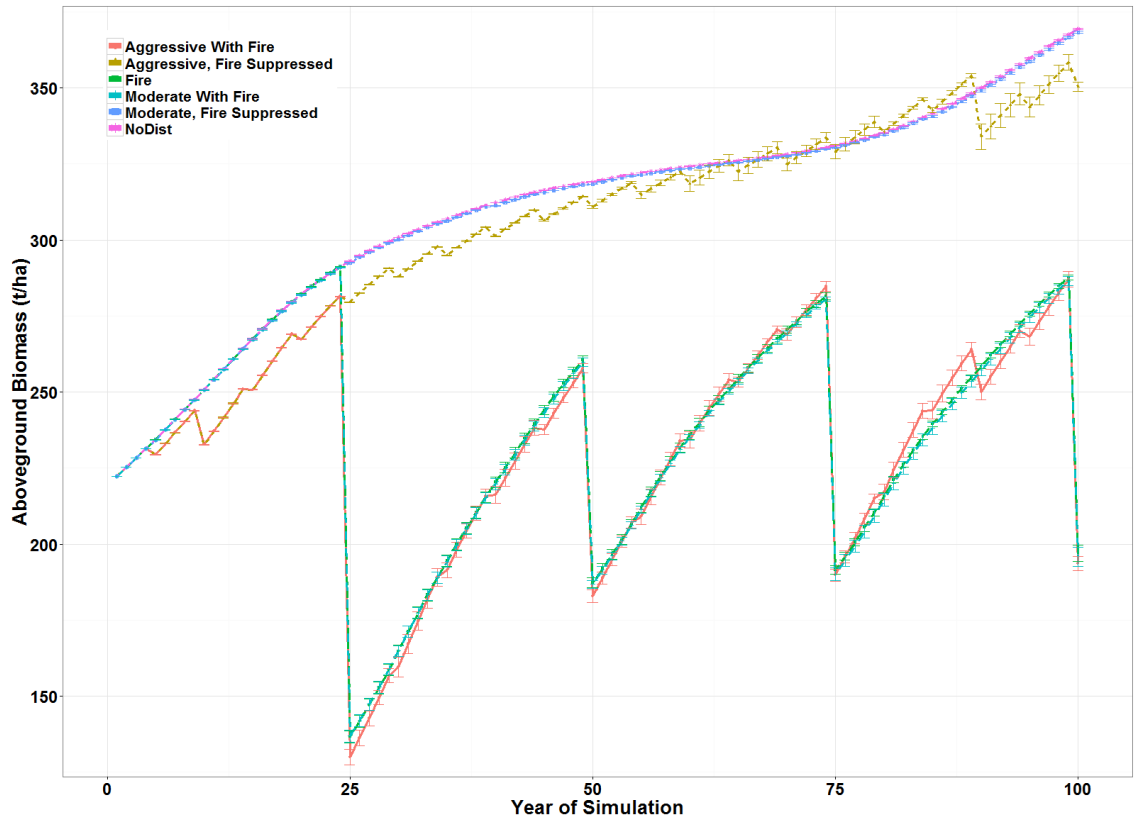


Figure 3.5: Change in biomass for different management regimes for the entire Grandfather Ranger District. The error bars indicate the 3 standard deviation of the 5 repeated model runs for each management scenario.

Moderate regime harvested 36.7%/ha/yr more sawlog and 32.7%/ha/yr more pulpwood than in the Aggressive regime. When fire is not suppressed, the Moderate regime harvested about 14.0%/ha/yr more sawlog and 6.3%/ha/yr less pulpwood than in the Aggressive regime. However, when harvested area is considered, the Moderate regime only produce 9% of the sawlog and pulpwood harvested in the Aggressive regime when fire is completely suppressed; and 7.2% of the sawlog and 5.9% of the pulpwood harvested when fire is not suppressed.

Another interesting result is that the Aggressive regime has more roundwood harvested when fire is not suppressed than when fire is completely suppressed: 17.7%/yr more sawlog and 14.6%/yr more pulpwood harvested. However, the Moderate regime behaves the opposite: 3.3%/yr less sawlog and 21.1%/yr less pulpwood when fire is not suppressed compared with than when fire is completely suppressed.

The time series of roundwood harvested per hectare (figure 3.6) shows that the sawlog harvested density in the Aggressive regimes fluctuate less than those in the Moderate regimes. Another noticeable thing is that the roundwood harvested per hectare in the Aggressive regimes exhibit a general decline over time, except for the peak in sawlog harvested around year 90 when no fire suppression. There is a sharp increase in sawlog harvested per hectare in the Moderate regimes in between year 22 and year 60. Pulpwood harvested per hectare in the Moderate regimes started out as about half of those in the Aggressive regimes, but the amounts become comparable to the Aggressive regimes after year 30 and suppressed those in the Aggressive regimes after year 80.

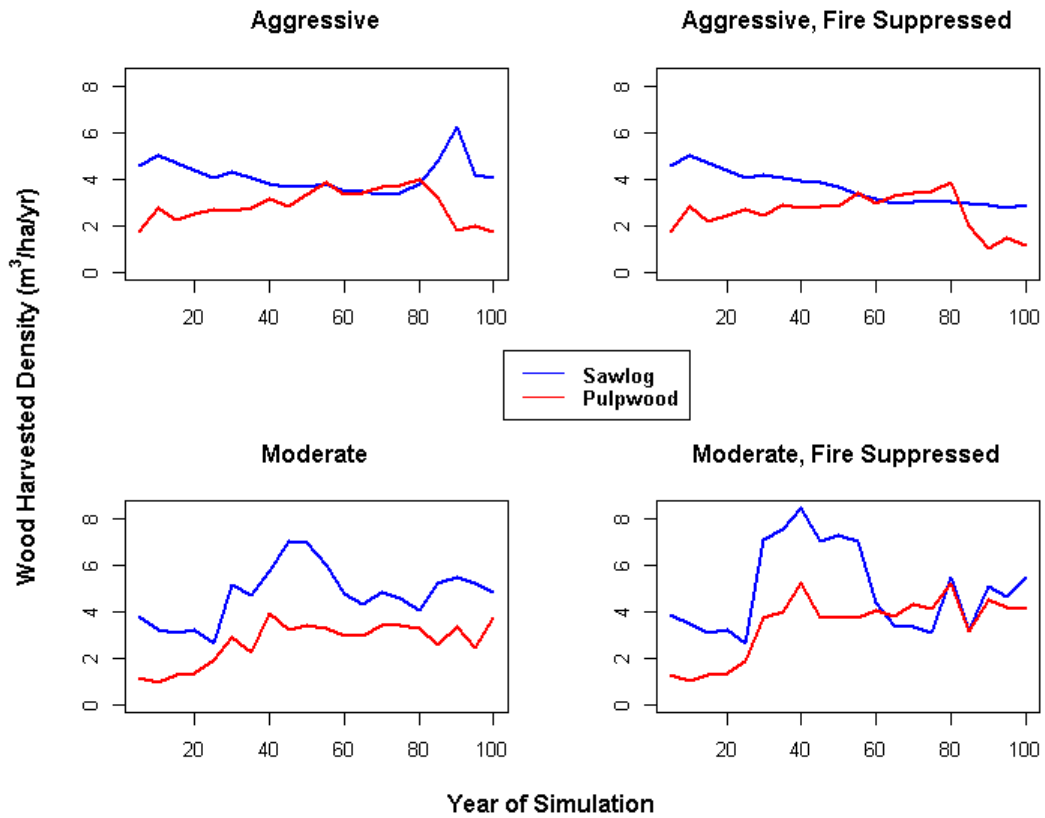


Figure 3.6: Time series of the sawlog and pulpwood harvest density for different management regimes for the entire Grandfather Ranger District. When taken into account of the harvested area, the annual volume harvested in the Aggressive regimes are about 15 times higher than that in the Moderate regimes.

Effect of treatments on provision of sustaining biodiversity

To understand how well the habitat's provision in sustaining biodiversity, the forest species diversity as indicated by 1. the Shannon's diversity Index, 2. the relative number of species, 3. the average age, and 4. the variation of the age structure that is shown as the age standard deviation were examined.

The Shannon's diversity index values are higher when fire is not suppressed. When fire is completely suppressed, the Aggressive regime result in a higher Shannon's diversity index than those of the baseline and the Moderate regime, but when fire is not suppressed, the index resulted from the Aggressive regime is the lowest (table 3.4). The student t statistics are all bigger than 1.96, which means that the difference in the Shannon's diversity index as the result of different management regimes, although varies, are significant (table 3.5). In particular, when fire is not suppressed, the Moderate regime is 12% more diverse than the same regime with complete fire suppression, but the Aggressive regime is only 1% more diverse when there is no fire suppression than the same regime when the fire is completely suppressed. Except for the Aggressive regime in the case when there is no fire suppression, which results in the landscape to be 3% less diverse than the baseline, all the other harvest regimes result in a more diverse landscape than the baseline.

The frequency distribution of species at the end of 100 years of the simulation for different management regimes (figure 3.7) shows that the Moderate regimes do not change the species ranking that is based on frequency compared to the baseline

Table 3.5: Comparison of the effective numbers derived from the average Shannon’s Diversity Index of the 5 model runs for the entire Grandfather Ranger District. The numbers on the top are the percentage difference of the effective number of species between the management regimes listed in the first column and those in the first row (the base). The numbers at the bottom with parentheses are the student t-statistics values when comparing the Index of the corresponding pairs of regime: “NoDist”=No disturbance; “Fire”=Fire only; “AgFire”=Aggressive with Fire; “AgNF”=Aggressive with complete fire suppression; “MdFire”=Moderate with Fire; “MdNF”=Moderate with complete fire suppression.

	NoDist	MdNF	AgNF	Fire	AgFire	MdFire
NoDist	0 (—)					
MdNF	2% (32)	0 (—)				
AgNF	16% (288)	6% (253)	0 (—)			
Fire	30% (412)	11% (384)	5% (194)	0 (—)		
AgFire	20% (291)	7% (262)	1% (59)	-3% (117)	0 (—)	
MdFire	33% (449)	12% (420)	6% (234)	1% (33)	4% (151)	0 (—)

scenarios. In the case of complete fire suppression, the Aggressive regime has a higher equitability of species distribution and higher percentage of the ecologically preferred species than the baseline and the Moderate regime. However, it also has higher proportion of invasive species. Similarly, when fire is not suppressed, although the equitability of species distribution is similar for all management regimes, the Aggressive regime still has a higher proportion of both ecologically desirable and invasive species when compared with the baseline and the Moderate regime. In contrary, although the Moderate regimes actively remove the 3 invasive species, their proportion is not that different from that in the baseline scenarios. Because the 3 most commercially desirable species (LITU, PIST, QUAL) were intensely planted in the privately owned forest and the 3 most ecologically desirable species (PIRI, QUAL, TSCA) were intensely planted in the publicly owned forest for the Aggressive regimes, they all have higher proportion than those in the baseline scenarios and the Moderate regimes. The higher proportion of the more desirable species, both commercially or ecologically, is the result of active planting more than of the opening up of canopy due to clearcut. Fire also opens up canopy, but based on the results of the baselines and the Moderate regime, opening up canopy does not really change the ranking of the four most abundant species. Additionally, the other 2 commercially desirable species (QURU and QUVE) that are not actively planted have much lower proportion than those that were actively planted. Although QURU (Northern Red Oak) was planted occasionally in the Moderate regime, it only yields slightly higher frequency than that in the baseline scenarios and the Aggressive regimes. The

introduction of more invasive species and lowering the average and the standard deviation of age of forest species agrees with field observation of the clearcut forest in the area (Leopold et al., 1985).

Base on the species distribution and the Shannon's diversity index, it seems that the Aggressive regimes provide a more desirable habitat for sustainability. However, if the age distribution of the trees are also considered, the average age of the forest in the Aggressive regimes is about 15 years younger when compared to the baseline scenarios and the Moderate regimes (table 3.4). The trees in that regime also have a smaller standard deviation of age, which means that the habitat under the Aggressive regime does not result in trees of different ages that can provide different niches for various animal species. The more desirable species distribution and the higher Shannon's diversity index in the Aggressive regimes may just be reflecting the condition of the forest when it is in the earlier forest succession stage.

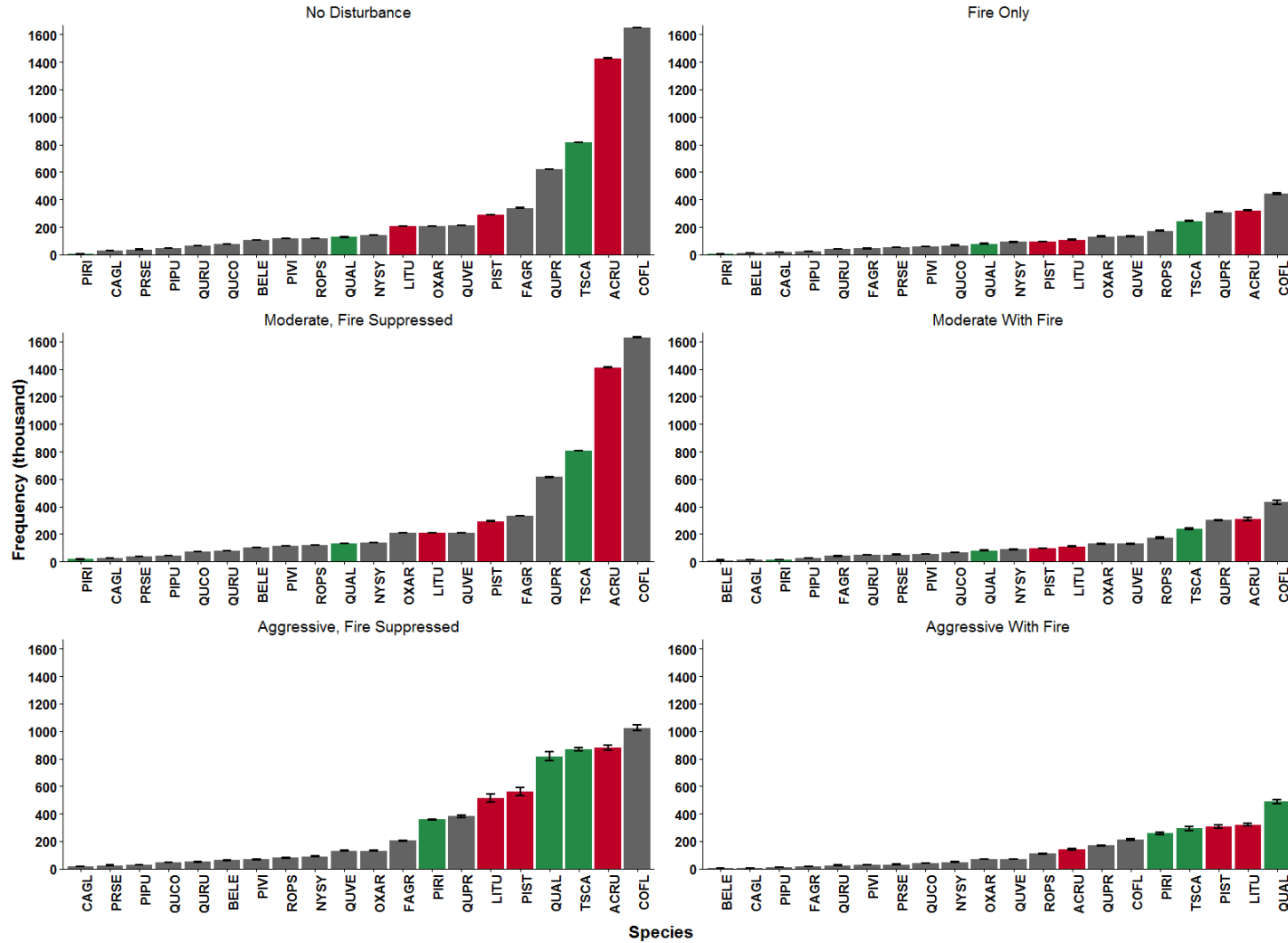
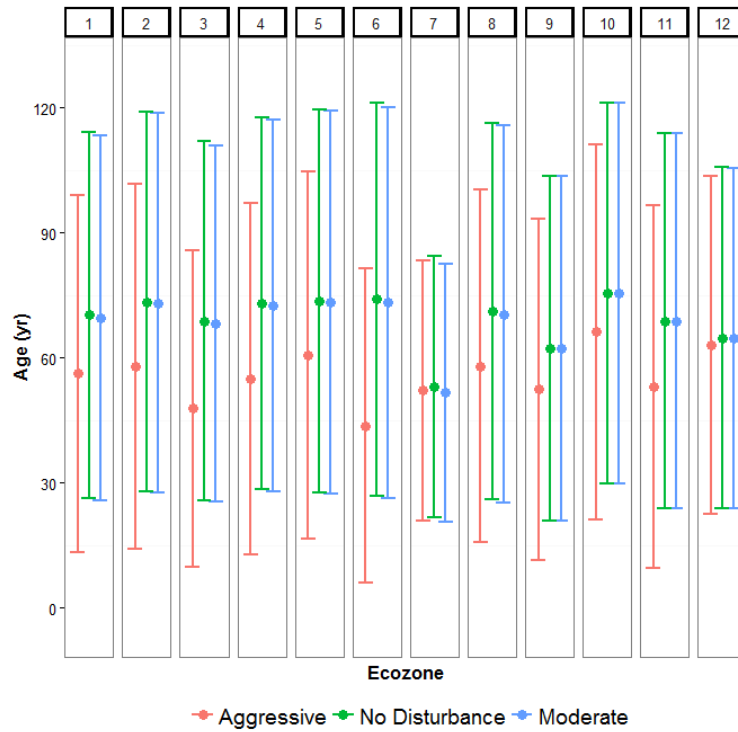


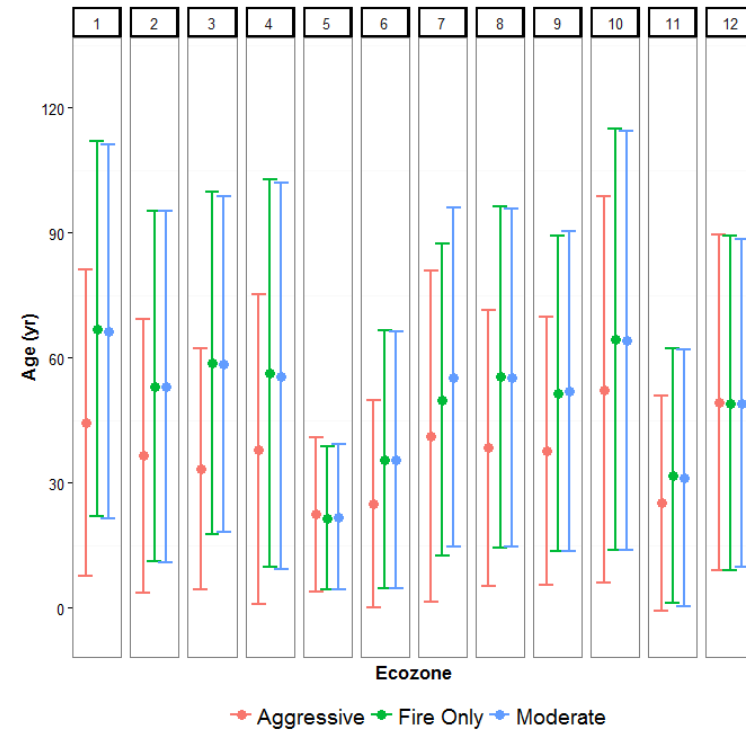
Figure 3.7: Frequency distributions of species 100 years after each management regime. Panels on the left column are management regimes with complete fire suppression, while those on the right are management regimes without fire suppression. Bars in red indicate the 3 most ecologically undesirable species, while bars in greens indicate the 3 most ecologically preferable species. Each bar indicates the average frequency of a species for running the models for 5 times. Error bars indicate 3 standard deviations of the mean for the 5 model runs, which accounted for 99.7% of the variations and visually more visible.

3.3.2.3 Spatial pattern of the effect of treatments

Effect of the management schemes on the species age and the standard deviation of age also vary by ecozone (fig. 3.3.2.3). The average age and the standard deviation of age for the Aggressive regimes are the lowest when compared with regimes of the corresponding fire status, but ecozones 2,3,4,5,6,11 are particularly sensitive. Specifically, ecozones 2,4, and 11 are suffered from having trees that are evenly aged (U.S. Department of Agriculture, Forest Service National Forests in North Carolina, 2011). For ecozones with special concerns (ecozones “Acidic Cove”, “Pine-Oak Heath”, “White Pine-Oak Heath”, “Rich Cove” and, “Shortleaf Pine-Oak”: ecozones 4, 5, 6, 10, and 11 in fig. 3.3.2.3), the Moderate regimes would result in older average tree age in those ecozones. Ecozones 5 (“Pine-Oak Heath”), 6 (“White Pine-Oak Heath”), and 11 (“Shortleaf Pine”) are especially sensitive to fire regime, where the average and standard deviation of age are much lower if fire is not suppressed.



(a) Complete fire suppression



(b) No fire suppression

Figure 3.8: Average and standard deviation of species age in each ecozone for different management scheme. Each code corresponds with each ecozone as follows: 1. “Northern Hardwood”; 2. Mesic Oak-Hickory”; 3. Oak Heath”; 4. “Acidic Cove”; 5. “Pine-Oak Heath”; 6. “White Pine-Oak Heath”; 7. “Spruce-Fir”; 8. “High Elevation Red Oak”; 9. “Unclassified”; 10. “Rick Cove”; 11. “Shortleaf Pine-Oak”; 12. “Dry Oak-Hickory”.

3.4 Discussion

Ecosystem services trade-offs

This study quantifies the trade-offs between the ecological services of the forest as a carbon sink, provision of wood products, and habitat for organisms of all types. The importance of this was acknowledged in the most recent IPCC report ([Smith & Bustamante, 2014](#)). The two harvest regimes were implemented in this study: “Aggressive” that consists of various forms of clearcutting; and “Moderate” that consists of various forms of species selection harvest that closely mimics the harvest regimes of the US Forest Service in the public lands. The Aggressive regime was a milder form of harvest prescription compared to the historical clearcutting in the Appalachian regions ([Yarnell, 1998](#)). Thus, the Moderate regimes can be viewed as restoration management regimes. This study is an example of scientific understanding of restoration actions on forest ecosystem and quantification selected ecosystem services trade-offs, which has been noted by [Benayas et al. \(2009\)](#) as relevant to conservation science.

Although the amount of roundwood harvested in the Aggressive regimes is larger than that in the Moderate regimes, the amount of harvested sawlog and pulpwood per hectare in the Aggressive regimes exhibit slight decreasing trends over time, but those for the Moderate regimes are slightly increasing (figure [3.6](#)). The larger amount of roundwood harvested in the Aggressive regimes is due to more areas

being harvested. The working hypothesis is that clearcutting is more cost effective than selective harvest (Keenan & Kimmins, 1993). However, the slowly decreasing trends may be an indication of the decrease in productivity and tree quality over time for the Aggressive regime, which is a sign that clearcutting may not be able to sustain a long-term high quality roundwood production. The decreasing trends for the roundwood harvested density would be higher if the clear-cutting areas per year is higher and occurs more frequently.

One of the advantages of using a landscape scale mechanistic model in a species-rich area, such as in this study, is that some aspects of habitat diversity can be calculated. While only the contribution of the forest trees to diversity is simulated, many other plant and animal species depend on, and differentiate between, the different tree species, their density and sizes. Here, the number of species, the numbers of individuals of each species, the age distribution and ecological importance rank were used to explore various aspects of diversity. These three are important characteristics that can complement one another in determining the quality of the habitat and biodiversity when evaluating ecosystem services trade-offs (Farnsworth et al., 2015). Specifically, in this study, although the Aggressive regimes result in the higher percentage of ecologically desirable species and higher Shannon's diversity index, other results show that the regimes also yield lower number of individual trees, and the average and standard deviation tree age overall (see table 3.4 and figure 3.7). The spatially explicit nature of the model also shows certain ecozones are more vulnerable to fire and the Aggressive regime (figure 3.3.2.3), especially for

some of them that require ecological restoration. The conclusion is that despite clear-cutting yields more roundwood harvested and higher proportion of ecological desirable tree species, the resulting forest would not be able to provide as many different ecological niches for various species as that resulting from the management regime that consists of various forms of species selection harvest.

3.4.1 Implication on conservation policy

Having a spatially explicit and qualitative information on habitat provision for sustaining biodiversity can help forest managers design a management regime that can balance the trade-offs between the ecological services of the forest as a carbon sink, provision of wood products, and habitat for organisms of all types for different parts of the forest (Benayas et al., 2009). Depending on the desired outcome, policy makers can provide subsidies or increase taxation to encourage forest managers to switch the forest management regime Masek et al. (2011).

Also because the results are spatially explicit, they can be used to inform forest management planning. For example, the Nantahala and Pisgah National Forests Land Resource Management Plan that is currently in the drafting process that describes the desired condition in each ecozone (National Forests in North Carolina, USFS, 2016). For example, the study results agree with the report that in the Pine-Oak Heath (ecozone 5) and Shortleaf Pine ecozones (ecozone 11), fire suppression determines the tree age structure, but are able to provide quantitative information on those selected ecosystem service trade-offs. The study results (figure 3.3.2.3)

provide the ranges of possible tree age and its variation as a result of complete and no fire suppression, which could be used to inform the forest manager on how much to control burning in those ecozones.

The model inputs used in this study assume that the temperature and atmospheric CO_2 concentration in this study only fluctuate within the level of the past 50 years. The future increase in the values of these two variables may increase tree growth, and possibly fire size and frequency, but these are beyond the scope of this study. Additionally, thinning is not simulated in this study, although it is a very common practice that allows the trunk of desirable trees to increase faster and to maximize the profit (Keyser & Brown, 2014). Accounting for the carbon flux to the harvested wood product pool from thinning and the impact of climate change coupled with different management regimes in a species rich area could be included in a future study. Nevertheless, the framework in this study provides forest managers a comprehensive spatially explicit information to understand the impacts of different management regimes on the trade-offs of the selected ecosystem services. The results can also be combined with other models to understand the impact on, for examples, animal diversity and landscape aesthetic.

Chapter 4: Mapping revenue from forest carbon subsidies and round-wood sales for different management regimes—with biophysical processes included

Abstract Carbon subsidies is one of the policies that governments are using to encourage forest landowners to manage their forest not just for maximizing timber harvest, such as the REDD+ program. Utilizing maps to perform scenario analysis of possible revenue of different landowners as a result of various environmental subsidies has been used to understand the effects of ecosystem service valuation policies. This study shows paying foresters at $US\$0.75(tCO_2e)^{-1}$, which is way lower than the social price of carbon, is an enough incentive for them to manage the forest for carbon sequestration. Even if subsidies are paid on the number of important species present, to maximize the revenue, foresters would still adopt some form of clearcutting regimes, but maybe plant more important species. The lessons are 1. using the social costs as the basis for subsidizing forest management may not be the most effective carbon subsidies to the foresters in terms of encouraging them to manage their forest for carbon sequestration; and 2. the subsidy of preserving the forest carbon in the biomass has to be higher than that of the annual carbon

sequestration if the goal of the policy makers is to prevent clearcut.

4.1 Introduction

Under the pressure of population growth and the increasing demand on ecosystem services, one of the ways to satisfy both conservation needs and the livelihoods of local people is by valuation of ecosystem services (Daily et al., 2000). Timber production and carbon sequestration are two of the ecosystem services that a forest provides. Timber production results ecosystem goods that is a manufactured capital, so the value of timber production could be fully captured in market. However, carbon sequestration is a regulating service that its value is not fully captured in market in terms comparable to other manufactured goods (Costanza et al., 1997; Nelson et al., 2008; Nelson & Daily, 2010). As a result, carbon sequestration, as public goods, is not commonly considered by forestland managers (De Groot et al., 2010).

To encourage forest owners to manage their forest not just for maximizing profit by harvesting timber, governments and organizations are starting to implement policies on providing subsidies on ecosystem services that are not commercially valued. The most common ones are subsidies on carbon. Programs such as the United Nations Collaborative Programme on Reducing Emissions from Deforestation and Forest Degradation (REDD+) (UN-REDD Programme, 2016) includes developed countries paying developing countries to preserve their forests or the implementation of Cap-and-Trade Program in California that provides the forest

owners (mostly within the United States) credit for carbon offsets ([California Environmental Protection Agency Air Resources Board, 2016](#)). However, some studies shows that subsidies that are solely based on carbon sequestration may have negative impact on the tree species diversity, such in the case of REDD+ ([Phelps et al., 2012](#)).

As carbon crediting for forest owners becomes more popular as a mean to mitigate climate change, numerous research has been conducted to understand the impact of different carbon prices and interests rates on the revenue of the foresters under different management scenarios. Current research is mostly either based on bioeconomic models that ignore the complex dynamics of forest growth and succession (e.g. [Caparrós \(2009\)](#)), or based on single tree species that cannot account for species competition during the succession process (for examples, [Buongiorno et al. \(2012\)](#); [Niinimäki et al. \(2013\)](#)). None of these research is spatially explicit, which means that the results are not applicable to most forest landscapes in the world. In addition, having the information of the species composition of each management scheme allows policy makers to understand the impact of the carbon credit on the habitat, if foresters adopt the management scheme that maximizes their profit based only on revenue from carbon credit and roundwood product selling. There is also a demand of accurately mapped and validated of land cover to understand the impact of ecosystem services valuation ([Foody, 2015](#)).

The physical scale of the ecosystem function and the scale at which humans value the goods and services provided are not necessarily the same ([De Groot et al.,](#)

2010, 2002; Hein et al., 2006). Environmental management activities such as timber harvesting, and the broad-range effects such as natural disturbances on a forest, are mostly concentrated at the landscape scale (Müller et al., 2010; Scheller & Mladenoff, 2007), which is a scale that consists of various land cover types and ecosystems that include human activities and natural disturbances. Its precise resolution and extent vary depending on the research question (Turner, 2005; Turner et al., 1989). . Landscape ecology can be used to address this problem because it explicitly links ecological processes and management (Mladenoff, 2004).

The objective of this paper is to study the ecosystem service trade-offs with the integration of forest succession processes at different carbon prices to understand the impact of different forest management regimes on the revenue of foresters and the number of individuals of the important and the invasive species. A spatially-explicit forest succession model, Landis-II (Scheller et al., 2007), has been employed in a species rich temperate forest to understand the impact of different management regimes on selected ecosystem services trade-offs at 150m resolution (Chapter 3, this dissertation). Two groups of carbon prices were used in this study: the social cost of carbon, and the market price of carbon, with different interest rates, to understand which carbon prices and regimes would provide different forest owners the maximum revenue.

4.2 Methods

4.2.1 Study area and management scenarios

The area of study was the Grandfather Ranger District (GRD) located in the Southern Appalachian Mountains in western North Carolina, USA. The area is characterized by the rapid change of terrain and diverse in tree species. The area is divided into 4 regions based on the management goal and the land ownership (figure 4.1). The 4 districts are: publicly-owned forest (“public”), privately-owned forest (“private”), ecologically important habitats (“ecology”), and Congressionally Designated Roadless (CDR) area (“roadless”). The CDR is area where no harvest or any management activity can take place. The management scenarios consist of two harvest regimes: “Aggressive” and “Moderate” (table 3.2), which the “Aggressive” regime consists of various level of clearcut and the “Moderate” regime consists of various level of selective harvest. The variation in harvest size and species depends upon the district where the area belongs to.

20 tree species were included in this study (table 3.1), with 3 of them are ecologically important (Class A) and another 3 are invasive (Class D).

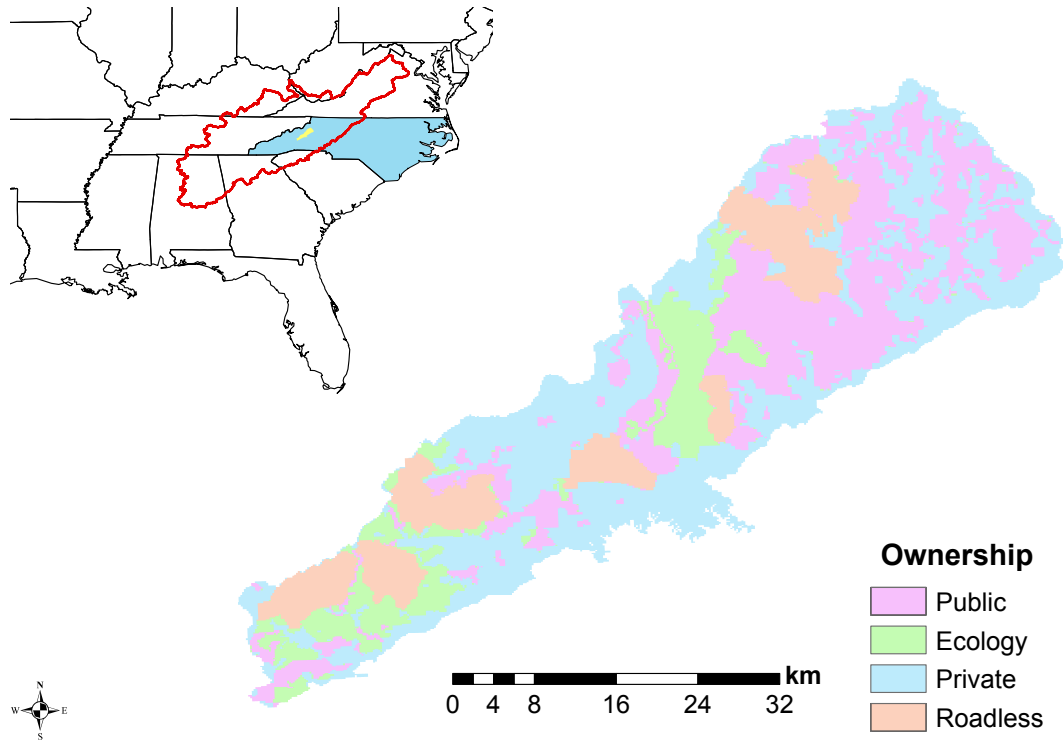


Figure 4.1: Forest ownerships in the Grandfather District (highlighted in yellow), North Carolina, USA. Red boundary indicates the Southern Appalachian Region.

4.2.2 Calculation of revenue

Revenue of the foresters in this case comes from both forest carbon credits they received from the carbon sequestration of their forest and from selling the lumber harvested from forest. Currently, the forest carbon trading market that is going to be implemented is the California Cap-and-Trade Program, which uses the U.S. Forests Compliance Offset Protocol to support the increased carbon sequestered in forests ([California Environmental Protection Agency Air Resources Board, 2015](#)). Carbon credits are paid to the forest owner in two aspects: “permanence” and “additionality”. “Permanence” is the carbon stock of a forest that can be maintained

100 years after the base year; “Additionality” is the additional carbon sequestered in the forest due to the growth of biomass. The unit of the carbon credit is converted from dollar per tonne of carbon-dioxide-equivalent to dollar per tonne of carbon to calculate the carbon stored and sequestered by the forest. Year 2000 was established as forest carbon baseline. In this study, the credit is not given to the carbon stored in the long-term roundwood products. The net present values (NPV) of all the revenue are reported based on 2015. i.e. the future revenue worths less in the present due to discount rate. Base on the area of each management region, the total revenue is normalized to \$/ha.

4.2.2.1 Revenue from selling wood products

The revenue for selling the wood product is based on the stumpage prices of sawlog and pulpwood([Extension Forestry](#), [Extension Forestry](#)). The units of the selling prices are converted to $\$/m^3$ from $\$/thousandboardfeet$ and $\$/cord$ using the conversion factors available in [UNECE \(2010\)](#). The present selling prices were calculated based on the average of the real stumpage prices of both sawlog and pulpwood in western NC from 2005 to 2015. This is a simple assumption that the real stumpage prices in the recent 10 years have remain relatively stable. The future selling prices are assumed to be fixed. The real prices for sawlog, and pulpwood were obtained by using the not seasonally adjusted producer price indices of “Lumber and wood products” (WPU08) and “Pulp, paper, and allied products” (WPU09) respectively to adjust for the impact of inflation. The data can be obtained from the

Bureau of Labor Statistics ([Bureau of Labor Statistics](#), [Bureau of Labor Statistics](#)). The average prices for sawlog and pulpwood are US\$204.4/m³ and US\$16.5/m³ respectively. The prices are reported in 2015 dollar-value.

4.2.2.2 Carbon credits and discount rates

The social costs of carbon and the market prices of carbon were used in this study. The estimates of the social cost of carbon varies. [Rogelj et al. \(2013\)](#) estimates the probability distribution of the social cost of carbon (in 2012 value) associated for specific future temperature increase targets. The more probable social cost is between US\$20 and \$40 per tonne of carbon-dioxide-equivalent emissions [$(tCO_2e)^{-1}$]. The social costs of carbon (in 2015 value) provided by the US EPA for annual interest rates of 5%, 3%, and 2.5% are US\$10 $(tCO_2e)^{-1}$, US\$36 $(tCO_2e)^{-1}$, and US\$56 $(tCO_2e)^{-1}$ respectively ([Interagency Working Group on Social Cost of Carbon, United States Government, 2015](#)). The social costs of carbon (in 2015 value) used in this study are US\$15 $(tCO_2e)^{-1}$, US\$45 $(tCO_2e)^{-1}$, and US\$150 $(tCO_2e)^{-1}$.

The other set of carbon prices used was is the carbon price in a voluntary carbon exchange market– the Chicago Climate Exchange (CCX). About 2 years before the end of the CCX market, the carbon price had dropped below US\$1 $(tCO_2e)^{-1}$ ([Intercontinental Exchange, 2016](#)). The 4 market prices used in this study are: US\$1 $(tCO_2e)^{-1}$, US\$0.75 $(tCO_2e)^{-1}$, US\$0.5 $(tCO_2e)^{-1}$. and US\$0.25 $(tCO_2e)^{-1}$.

There are much debates in the discount rate of carbon–see [Stern \(2007\)](#), [Nordhaus \(2007\)](#), and [Dietz & Stern \(2008\)](#) for examples. People who think long term

issues such as climate change caused by carbon emission should have a lower discount rate, while others who believe in technological breakthrough in the near future think the discount rate should be higher. The annual discount rates used in this study are 0%, 1% and 5%.

4.3 Results

4.3.1 Revenue

4.3.1.1 How do different carbon credits affect revenue?

Results were analyzed by comparing the impact of the carbon credit and discount rate, the method of management, fire suppression scheme, and the nature of the land ownership on the revenue. In general, the proportion of revenue from selling carbon credit is higher than that from selling the harvested wood, except for the privately-owned forest in the Aggressive regimes. Also, for the market prices of carbon, some form of harvesting always results in high revenue than not harvesting at all, as indicated by the CDR always has the lowest revenue regardless of management regime (table 4.1).

Even when the carbon credit is $\text{US}\$150(tCO_2e)^{-1}$, only in the Moderate regime with no fire suppression, the revenue in the CDR is higher than the privately owned forest, almost equal to that for the public forest, but still less than that for the ecologically sensitive area that still conduct a minimal amount of harvest (table 4.2).

Table 4.1: NPV revenue (k\$/ha) at different management regimes and regions of management (“Roadless” indicates the Congressional Designated Roadless area where no human activities occur. The first row indicates the carbon credit (in 2015 US dollar per tonne of carbon-dioxide-equivalent emissions). “Ag-Fire”=Aggressive With Fire; “AgNF”=Aggressive with complete fire suppression; “MdFire”=Moderate with Fire; “MdNF”=Moderate with complete fire suppression.

Management	Region	$\leq \$0.50$			\\$0.75 and \$1		
		Total	Carbon	Wood	Total	Carbon	Wood
AgFire	Public	15.1	9.3	5.8	27.5	21.6	5.8
AgFire	Ecology	9.8	8.4	1.4	21.0	19.6	1.4
AgFire	Private	26.0	10.6	15.4	40.2	24.8	15.4
AgFire	CDR	7.8	7.8	0.0	18.3	18.3	0.0
AgNF	Public	12.1	5.4	6.7	19.4	12.7	6.7
AgNF	Ecology	5.6	4.2	1.4	11.1	9.7	1.4
AgNF	Private	25.1	7.4	17.8	35.0	17.2	17.8
AgNF	CDR	3.4	3.4	0.0	8.0	8.0	0.0
MdFire	Public	8.2	7.8	0.4	18.6	18.2	0.4
MdFire	Ecology	8.2	7.9	0.3	18.7	18.4	0.3
MdFire	Private	7.2	7.1	0.1	16.6	16.5	0.1
MdFire	CDR	7.8	7.8	0.0	18.2	18.2	0.0
MdNF	Public	4.1	3.7	0.5	9.0	8.6	0.5
MdNF	Ecology	4.0	3.6	0.3	8.8	8.5	0.3
MdNF	Private	3.6	3.5	0.1	8.4	8.2	0.1
MdNF	CDR	3.4	3.4	0.0	8.0	8.0	0.0

Table 4.2: Comparing revenue of other forest management regions with that at the Congressional Designation Roadless (CDR) region where no management activity occurs. “AgFire”=Aggressive With Fire; “AgNF”=Aggressive with complete fire suppression; “MdFire”=Moderate with Fire; “MdNF”=Moderate with complete fire suppression.

Management	Region	Revenue (k\$/ha)
AgFire	Public	3717
AgFire	Ecology	3365
AgFire	Private	4266
AgFire	Roadless	3134
AgNF	Public	2176
AgNF	Ecology	1663
AgNF	Private	2960
AgNF	Roadless	1372
MdFire	Public	3113
MdFire	Ecology	3156
MdFire	Private	2821
MdFire	Roadless	3113
MdNF	Public	1469
MdNF	Ecology	1456
MdNF	Private	1413
MdNF	Roadless	1375

4.3.1.2 Spatial patterns of revenue

The spatial pattern of the revenue from different management regimes using carbon credit of $\text{US\$}45(t\text{CO}_2e)^{-1}$ and the annual discount rate of 1% is shown in figure 4.2. The Moderate regimes result in less difference in revenue for different management regions. Such kind of spatial map may be of interest to forest owner on the ground. For instance, the private forest on the northeastern edge and that on the middle of the northwestern edge of the Grandfather Ranger District may be interested to know that if they practice the Aggressive regime even with complete fire suppression yields higher revenue than the Moderate regime with no fire suppression.

In the privately -owned forest located on the middle of the northwestern edge of the Grandfather Ranger District, there are some patches within the forest that are publicly owned (figure 4.3). Some of the publicly owned forest patches receive equally high revenue compare to that particular privately owned forest in the surrounding, while some patches have obviously lower revenue. Further contact with the Forest Service is needed to determine whether the lower revenue in some of the publicly-owned patches is due to the difference in productivity or management regimes.

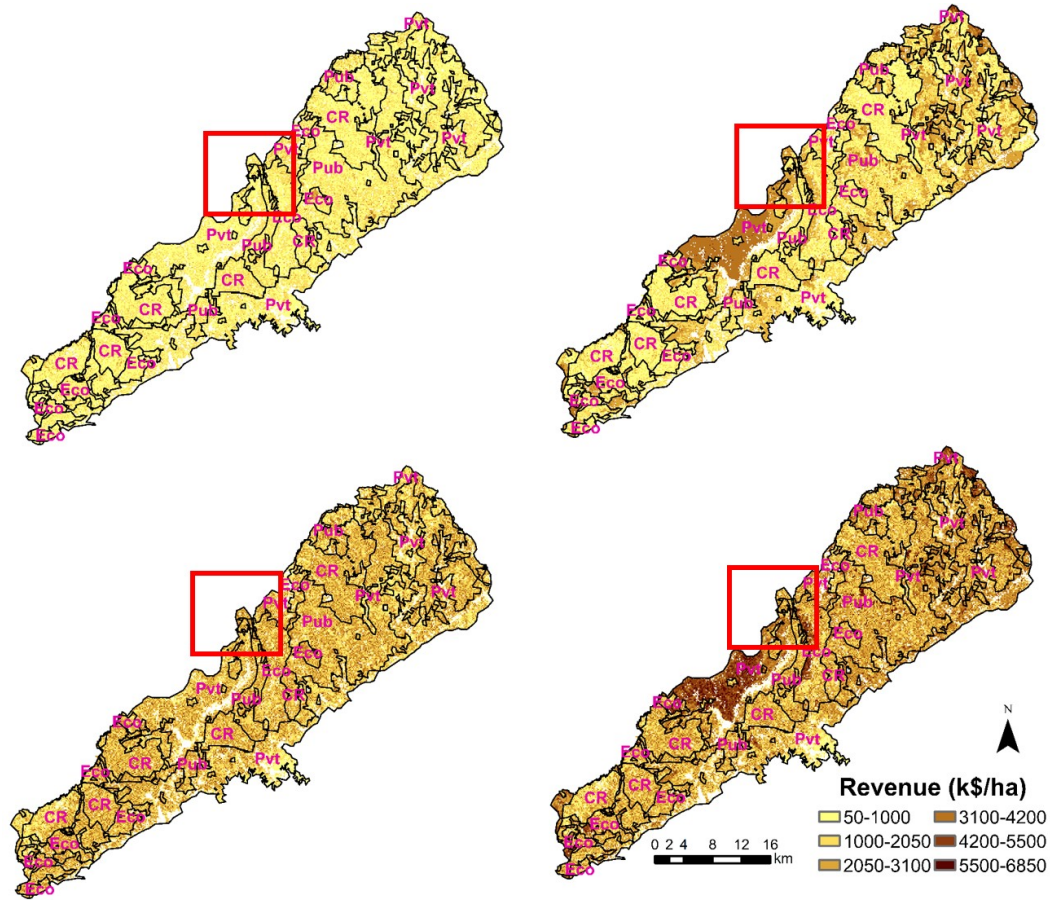


Figure 4.2: Spatial pattern of the forest owner revenue for carbon credit of $\text{US}\$45(t\text{CO}_2e)^{-1}$ and annual discount rate of 1, with harvest rotation of 80 years. The top row is the management scenarios when there is no fire suppression and the bottom row is when the fire is completely suppressed. Figures on the left column are the Moderate regimes and those in the right column is the Aggressive regime. Red boxes indicate the area that is going to be examined in more detail.

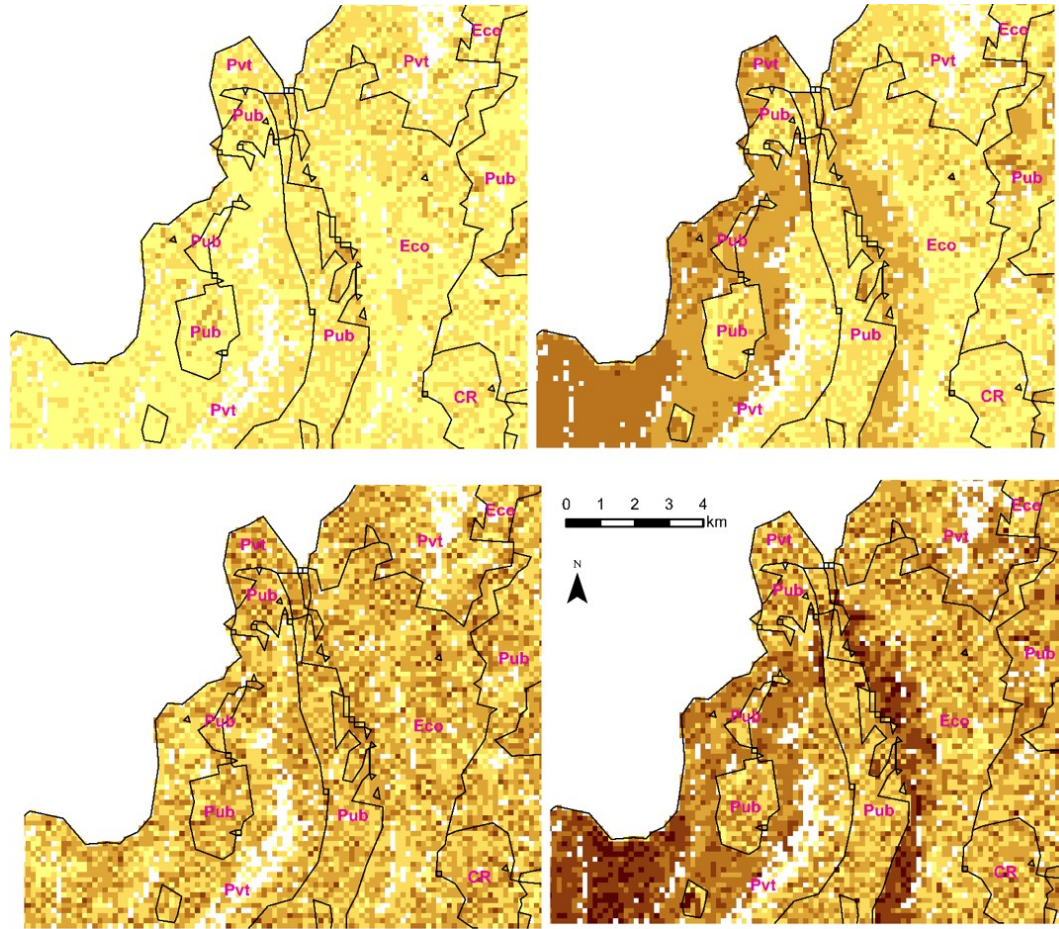


Figure 4.3: More detail analysis of the spatial pattern of the revenue at the selected regions from figure 4.2 (area in red box). The top row is the management scenarios when there is no fire suppression and the bottom row is when the fire is completely suppressed. Figures on the left column are the Moderate regimes and those in the right column is the Aggressive regime.

4.3.2 Important vs invasive species

The Aggressive regime generally yield more trees that belong to either Class A or Class D, except for the CRD where no harvest occur (table 4.3). Consider the CRD, if an area has no harvest or planting occur, fire suppression yields more than double of Class A and Class D compared with those when fire is completely suppressed. Regardless of management regime, the number of trees belong to Class D is more than those belong to Class A.

4.4 Discussion

4.4.1 Importance of the spatial resolution

Due to the species and topographic heterogeneity of the area, the revenue variation varies greatly depending on the area. This study, all the private lands were classified as one entity. However, in real life, private lands are often owned by different owners. The fine resolution estimate, at 150m, allows landowners on the ground to understand the potential impact of different policy on their revenue. Figure 4.3 shows an example of the heterogeneity nature of the revenue. If the study and the model results are conducted in a coarser resolution, then a lot of the difference in revenue earned will not be observed. Such analysis may be useful for policy makers at the state or national level, however, it lacks useful information for the local stakeholders, i.e. forest landowners, to understand how various carbon

Table 4.3: Number of trees (thousand/ha) belong to Class A or Class D presence for different management regime at different management region. “Ag-Fire”=Aggressive With Fire; “AgNF”=Aggressive with complete fire suppression; “MdFire”=Moderate with Fire; “MdNF”=Moderate with complete fire suppression.

Management	Region	Class A	Class D
AgFire	Public	19.4	4.7
AgFire	Ecology	12.1	6.1
AgFire	Private	9.3	15.9
AgFire	CRD	4.2	6.4
AgNF	Public	26.8	15.1
AgNF	Ecology	18.6	15.2
AgNF	Private	13.0	23.0
AgNF	CRD	8.2	16.9
MdFire	Public	4.6	6.5
MdFire	Ecology	4.3	6.0
MdFire	Private	4.6	7.3
MdFire	CRD	4.1	6.3
MdNF	Public	8.5	16.4
MdNF	Ecology	8.3	16.6
MdNF	Private	8.0	16.0
MdNF	CRD	8.3	16.9

credit policies may impact them so that they can make a more informed decision to manage their forest. In addition, because the revenue calculation is based on a calibrated and validated detailed forest succession model that accounts for natural disturbances and species competition, the expected outcome will be more accurate than the results based on the carbon and roundwood production derived from a simple growth and yield model.

4.4.2 Implication on policies on carbon subsidy and forest habitat sustainability

Currently there are two systems in accounting for carbon credit, the social cost of carbon and the market price of carbon. The social cost of carbon accounts for the possible monetized damages as a result of different levels of increase in temperature (Greenstone et al., 2013), which is more than $\text{US\$}15(t\text{CO}_2e)^{-1}$, in 2015 US dollars. However, regardless of the carbon credit, the Aggressive regime with no fire suppression results in the highest revenue for the whole region. This management regime has the most forest disturbed compared to other regimes. It results in a less number of trees and the youngest average age of forest. Younger trees sequester more carbon compared to older trees. Under the carbon payment system used in this study, where the amount of subsidy provided to the additional carbon sequestered and that to the carbon that are retained in the forest biomass is the same, then one of the possible outcomes is that foresters may do more frequent clearcutting and grow

more invasive and fast-growing species to maximize their revenue on both selling the harvested roundwood and on selling the carbon credit.

The other interesting result is that unless the carbon credit is as high as $\text{US}\$150(t\text{CO}_2e)^{-1}$, then some form of harvesting still results in higher revenue in this case study. Since the NPV of the revenue from wood harvesting remains the same regardless of the change in carbon credit based on the assumption in this study, the proportion of wood harvesting revenue consists of only a small percentage of the total revenue when carbon credit is more than $\text{US}\$15(t\text{CO}_2e)^{-1}$. One of the possible reasons for the revenue at the CDR is almost always the lowest is due to the productivity of the area. Although the study area is relatively small and the CRD is geographically close to other management regions, the heterogeneity nature of the landscape shows that the proximity cannot be used to assume similar productivity of the forest. This also proves the importance of high resolution study in the heterogeneous and species rich areas.

[Rogelj et al. \(2013\)](#) finds that the probability of achieving the 2°C increase before the end of the century increase from 50% to 66% if the carbon price increases from \$20 per tonne of carbon-dioxide-equivalent emissions $[(t\text{CO}_2e)^{-1}]$ to $\text{US}\$40(t\text{CO}_2e)^{-1}$ with some limit on energy demand or some future technological breakthrough and if the cost are implemented right now; a carbon price of higher than $\text{US}\$150(t\text{CO}_2e)^{-1}$ to achieve the objective with a probability of more than 66% if there is no limit on energy use or delay in implementing to impose any cost on carbon. The result shows that paying the foresters more than $\text{US}\$0.75(t\text{CO}_2e)^{-1}$ is not

needed to encourage foresters to consider managing carbon for their forests. If the carbon subsidies to the foresters equal to the social cost of carbon, at $\text{US}\$40(t\text{CO}_2e)^{-1}$, the question would be, is 98% of that payment more than the overhead cost of accounting and monitoring carbon in a forest? If that is the case, then what could be a more efficient use of the excess money?

The results also show that the higher the discount rate, the lower the NPV of the total revenue. The analyze of the discount rate echoes with the comments of [Stern \(2016\)](#) that the discount rate is central to the discussion of climate change mitigation. Although [Rogelj et al. \(2013\)](#) considers carbon price of less than $\text{US}\$1(t\text{CO}_2e)^{-1}$ as “the absence of any serious mitigation efforts”, if the only other completing revenue is selling harvested roundwood, then paying foresters at $\text{US}\$0.75(t\text{CO}_2e)^{-1}$ is an enough incentive for them to manage the forest for carbon sequestration based on the result of this study. Future research needs to be done to understand what if the carbon subsidies also apply to the sawlog harvested and the interaction between roundwood prices and carbon subsidies. The number of trees in the important and invasive does not completely reflect the ability of a habitat to sustain biodiversity. Even if there are subsidies on the number of important species present, to maximize the revenue, foresters would still adopt the Aggressive management regimes while planting more ecologically important species. If the goal of the policy makers is to prevent clearcut, other indicators would have to be considered.

Chapter 5: Discussion

5.1 Major research questions answered

The dissertation research aims to advance systematic understanding of the combined impact of both natural disturbances and management on forest carbon dynamics, species diversity, and roundwood production in the context of improving carbon offset policy. Specifically, I answered the following questions:

1. *How do forest management activities contribute to the amount of harvested wood products?*

In chapter 2, an empirical model that links forest management activities with the amount of harvested wood product was developed. The results show that the eastern part of NC contributes the most of the roundwood produced in the state during 1986 to 2010. In particular, when forest area is taken into account, the northern Coastal Plain contributed the most to roundwood production.

2. *How to quantify the annual contribution of forest management activities to the carbon storage in different types of roundwood products at county level and why is it important?*

Remote sensing detected forest disturbance maps were combined with other data to develop a spatially explicit relationship between forest management activities and roundwood harvested (chapter 2). Previous approaches to estimating the carbon stored in the HWP provide estimates either at a national or regional level (see, e.g., [Apps et al., 1999](#); [Donlan et al., 2012](#); [Heath et al., 2011](#); [Karjalainen et al., 1994](#); [Masek et al., 2011](#); [Pan et al., 2011](#); [Stockmann et al., 2012](#); [Zhou et al., 2013](#)), or for specific forests ([Profft et al., 2009](#)) over a few years. Other studies do not provide the contribution of forest management activities ([Adams et al., 2006](#)) or detailed information on the types of roundwood produced ([Huang et al., 2015](#)) which is needed for further analysis of the dynamics of roundwood production and life cycle analysis, to improve accounting of long-term carbon stored. The latest IPCC Guidelines ([IPCC, 2014](#)) called for study for fine scale (1ha) level of study that links forest management activities to carbon flux to the HWP pool. Not only does it affect carbon credit trading between countries, but also forest carbon accounting in both the standing forest biomass and estimation of the carbon decay rate in the HWP pool.

3. *How are the carbon sequestration and species composition impacted by natural disturbances and various management regimes in a species-rich environment at landscape scale (~ 2ha resolution)?*

Chapter 3 shows that the fire regime influences the most on carbon sequestra-

tion and species composition than harvest intensity: Fire suppression result in the forest standing biomass acting as a carbon sink, and it creates more preferable tree age distribution in some ecozones. Harvest regime that consists of various clearcut yields most harvested roundwood, but results in the least amount of carbon sequestered by the forest standing biomass. However, carbon stores in the long term product pool (sawlog) for another 100 years. This harvest regime on the one hand lowers the number of trees, the average and the standard deviation of tree age in the forest, on the other hand increases the proportion of both ecologically important and invasive tree species. Its impact on the desirability of the habitat for wildlife species needs further research.

4. *How do different carbon credits and interest rates affect the revenue of foresters spatially?*

Results in chapter 4 show that spatial difference in revenue is more pronounced in the Aggressive management regimes or when fire is not suppressed (figure 4.2). Further investigation is needed to understand whether the spatial difference in revenue is influenced more by the productivity of a particular area or the levels of carbon credit.

5. *Which management regime would the foresters likely to adopt if they were to maximize their revenue?*

Results in Chapter 4 show that the management regime that consists of various clearcut and no fire suppression maximizes the revenue compared to manage-

ment regimes that consist of selective species harvest or scenarios when fire is completely suppressed. This can be explained by because clearcutting results in younger trees in the forest and higher amount of roundwood harvested. Revenue from both increasing carbon sequestration rate and selling more harvested roundwood is much higher than the revenue coming from retaining the forest biomass for 100 years. Although the proportion of revenue from selling harvested roundwood is low as the payment of carbon credit increase above $\text{US}\$0.75(t\text{CO}_2e)^{-1}$ (2015 dollars), performing some form of harvest always result in higher revenue per hectare than areas that have no harvest at all. This may be due to the difference in productivity of the forest owned by different entity.

5.2 General discussions

The dissertation research quantify the trade-offs between carbon sequestration, total timber harvested, and habitat provision for sustaining biodiversity, as well as to understand the revenue of the foresters at different location for various carbon payment scenarios. To achieve this goal, a research framework that combines remote sensing, statistical model, spatially explicit mechanistic forest succession model was developed.

General relation between forest management activities and annual production of sawlog and pulpwood at the county level is needed to provide general understanding of the spatial pattern, the trend of roundwood production, and the harvest

density in the area so that the impact of the factors that vary spatially, such as biophysical factors, prices, and disturbances, on roundwood production can be understood. It can also be used to compare the historical harvest pattern with that in the future simulated scenarios. For instance, the model simulated average annual harvested volume per hectare in chapter 3 (table 3.4) agrees with the harvest density of the forestland in the corresponding counties of the GRD in between 1986 and 2010. Figure 2.1(b) in chapter 2 shows that it is about $0.08m^3/ha/yr$ to $3.67m^3/ha/yr$.

Chapter 3 analyzes the trade-offs between carbon sequestration by forest biomass, sawlog and pulpwood harvested, species diversity, the proportion of each species in the landscape, the average forest tree age and the ability of the forest in providing different niches of habitats for different species, as indicated by the standard deviation of the forest tree age, for different management regimes. The last 4 served as indicators of the habitat in sustaining biodiversity. When fire is not suppressed, the aboveground forest biomass becomes a carbon source, which implies that fire suppression decision is a more important factor compared to the harvest regime adopted when determining whether the forest biomass would be a carbon source or a carbon sink. Harvest essentially removes some of the carbon that was originally stored in the biomass to various types of product pools: the IPCC Guidelines assume the turnover time of carbon stored in pulpwood is 3 years on average and that in sawlog is 100 years on average (IPCC, 2006). The two harvest regimes implemented in this study: “Aggressive” consists of various forms of clearcutting; and “Moderate” consists of various forms of species selection harvest that closely mimics the harvest

regimes of the US Forest Service in the public lands. The Aggressive regime is a milder form of harvest prescription compared to the historical clearcutting in the Appalachian regions (Yarnell, 1998). Thus, the Moderate regimes can be viewed as restoration management regimes. Results in chapter 3 shows that although the yield of harvested roundwood in the Aggressive regimes are higher than that from the Moderate regimes due to the area harvested is larger, the decreasing trend in the volume harvested per hectare over time shows that such method of harvest may not be able to sustain a long-term high quality roundwood production. In addition, although the Aggressive regimes do not result in the decrease in the proportion of ecologically important species, such regimes increase the proportion of ecologically undesirable species. Moreover, the decrease the average and standard deviation of tree age compared to the baselines and the Moderate regimes implied that the resulting forest landscape became less able in providing different ecological niches for species.

The results in chapter 4 shows that if the subsidy for retaining the biomass for 100 years is the same as that for the amount of annual carbon sequestered, then that may not be enough of an incentive to prevent clearcutting. Even if there is an additional subsidies on the number of ecologically important species presence, it will not prevent clearcutting. The carbon prices used in this chapters are the market prices, which are US\$0.25, 0.50, 0.75, and $1(tCO_2e)^{-1}$ (2015 dollars), and the social costs, which are US\$14, 45, and $150(tCO_2e)^{-1}$ (2015 dollars). When carbon credit is more than $US\$0.75(tCO_2e)^{-1}$ (2015 dollars), then the proportion of revenue com-

ing from selling the carbon credit is already much higher than that coming from selling the harvested roundwood. Various studies, for examples, [Rogelj et al. \(2013\)](#) and [Interagency Working Group on Social Cost of Carbon, United States Government \(2015\)](#), indicate the social cost of carbon is around $\text{US\$}20\text{-}\$40(t\text{CO}_2e)^{-1}$ (2015 dollars). A question would be whether paying the foresters at the social cost of carbon is efficient in encouraging foresters to manage their forest also for carbon sequestration. Results from chapter 4 show that the unintended consequence of carbon subsidize policy is that it seems to encourage more clearcutting, even if part of the carbon credit is paid for maintaining the same amount of carbon stored in the forest biomass in the same sites (at 150m resolution) for 100 years. Because clearcutting does not decrease the proportion of ecologically important species if they are planted, a credit on the presence of ecologically important species does not prevent foresters from clearcutting. Forest carbon credit payment programs such as the REDD+ and the California Cap-and-Trade program need to reconsider if equal credit should be given to carbon sequestration and retaining the same amount of carbon stored in forest biomass.

5.3 Implications

Chapter 2 demonstrates a statistical method that can quantify the annual relationship between forest management activities and the amount of roundwood harvested at the county level. In particular, the types of harvested roundwood was distinguished as sawlog and pulpwood, which sawlog is a long-term carbon pool of

about 100 years (IPCC, 2006). The research in chapter 2 established a framework that uses remote sensing images, forest inventory data, and land cover type map that links annual county-level forest management activities to the carbon flux to the long term pool in the harvested wood product (HWP), when combined with the allometric tables such as developed by Smith et al. (2006), can be used to estimate the amount of carbon flux to the long term HWP. The provision of a consistent fine scale information can improve the current method of carbon accounting (Zhu et al., 2010). It is also one of the important goals of the North American Carbon Program (Kasischke et al., 2013). Such method can also help countries to fulfill the newest IPCC estimation guideline (IPCC, 2014) to include the contribution of Harvested Wood Product (HWP) to the change of carbon stock in forests. It could help countries to develop domestic management policies aimed at meeting their carbon storage goals in both standing forest biomass and HWP. The annual time series maps of roundwood production can be further used to study the interaction between the management policies, change in roundwood prices, and the amount of roundwood harvested.

In a species-rich heterogeneous forest, a spatially explicit mechanistic forest succession model that can account for the biophysical factors of different ecozones, influences from natural disturbances, and species competition during the forest succession process is needed to provide an accurate quantitative estimate of ecosystem services trade-offs under possible new management regimes that may be used in the future (Gustafson, 2013). Additionally, chapter 3 demonstrates how such model,

which accounts for the characteristics and behaviors of different species, is needed to provide a more comprehensive view of the forest habitat, because it can provide multiple indicators that reflect the capability of forest habitat provision for sustaining biodiversity. The definition of the term “biodiversity” is ambiguous, as there is no single indicator that can represent the meaning of the term and the scientific definition of this term is often differ from what the majority of the public perceives (Farnsworth et al., 2015), so there has been difficulties in quantifying the quality of the habitat and resulted in a debate between ecologist and economist when trying to use a single indicator such as the number of species presence to value “biodiversity” (see Bartkowski et al. (2015) and Farnsworth et al. (2015) for example). Results from such detail-level models can help resolve some of the dispute in valuation of biodiversity.

Chapter 4 uses the result from chapter 3 to map the revenue for scenarios of different carbon prices and interest rates. It shows the revenue of the foresters under different carbon subsidies for different forest management scenarios. The analysis in chapter 4 is a step in advance in spatially explicit carbon and timber valuation tool, such as InVEST, as it included the forest succession processes in accounting the carbon sequestered and the timber harvested (Tallis & Polasky, 2011). In addition to revenue information, the result also include the impact on the forest habitat under different management regimes. Results in this chapter provide more than just the species presence, but also the age structure and the proportion of each species, can help further advance the study in biodiversity valuation because it allows further

cost-benefit analysis ([Farnsworth et al., 2015](#)).

The dissertation is the first step of research in developing a dynamic model that fully couples the social and forest ecological system. If the factors affect individual choices of harvest at the local level and those affect the supply of harvested timber at the regional level are understood, then a dynamic model that consists of management decisions, natural disturbance, ecosystem service valuation policies, and forest ecosystem response can be developed. Such model can help better understand the ecosystem services valuation policy on the financial well-being of foresters and the health of the forest ecosystem.

5.4 Limitations

The idea of ecosystem service valuation is often criticized as anthropocentric: human beings are valuing agents that translate the basic ecological structures and processes into values to help guide decisions ([De Groot et al., 2010](#)). However, valuation is not a solution or an end in itself: it is one of the tools in a larger political decision making ([Daily et al., 2000](#)). This dissertation work is the first step allowing us to develop a fully coupled model that takes into account of the interaction between the forest ecosystem and the possible change of the management regimes as a result of exogenous environmental policies such as the introduction of carbon price. The model can be applied globally at the landscape level. The possible changes in management practices for the forest landowners are based on very simple assumptions. For example, the cost of harvest is not accounted for and the future

roundwood real price is assumed to be constant. Also, land use or land cover change is not taken into consideration, which means the opportunity cost of the land is zero. In addition, to understand the decision process of the landowners, survey of the landowners' response must be conducted to simulate a more realistic response to the policy. Nevertheless, if more detailed social studies were done to adjust those assumptions, it will be possible to inform policy at much finer scales, approaching that of the individual forest managers while, at the same time, providing realistic scenarios that can be used by all the stakeholders.

5.5 Future directions

The following are proposed projects for further continuation:

1. *Create a dynamic model that fully couples the social and ecological system*

Further study can be done in understanding the factors affect individual choices of harvest at the local level and those affect the supply of harvested timber at the regional level.

2. *Include more ecosystem services and possibility of land-use change*

Besides habitat provision and carbon sequestration, forests also act to improve groundwater quality and attract tourists. Those additional ecosystem services can also be added to the model. In addition to managing the forest, the forest landowners could also convert the forest into other land-use, such as agriculture or urban. The possibility of land conversion can be factored into

the opportunity cost of the forest, which can also be incorporated in analyzing the possible response of the landowners.

3. *Examine the effect of scale*

Forest ecosystems and human agents are never homogeneous. However, policy on taxation or carbon subsidy usually come from a coarser scale. There has been no study that quantify the possible difference if the ecosystem dynamics is applied at a coarse resolution with details of the impact being averaged compared to if the ecosystem dynamics is analyzed at a landscape scale, like objectives 1-3 in section 1.2. Other spatial forest succession model, such as the Ecosystem Demography (ED) Model described in (Moorcroft et al., 2001), has been applied at the global scale, which can be used as a comparison to understand the effect of scale in analyzing the impact of ecosystem service valuation policy on the ecosystem and the economic benefits of the stakeholders.

4. *Application to data-poor areas*

One of the difficulties in ecological modeling is the limitation on data. To avoid more complication on the developing stage, the fully coupled model described would be applied to the forest systems in the United States as pioneer studies. A lot of ecological-sensitive forests are located in developing countries, where a lot of ecological data are not available. Moreover, forest landowners in the tropical developing countries will receive subsidy from developed countries to preserve their forest under the REDD+ scheme (Phelps et al., 2012).

Field work and remote sensing data would have to be incorporated to apply this model in data-poor areas. Nevertheless, a fully coupled spatially explicit ecological-economic model applied in developing countries can help assess the REDD+ program.

Appendix A: Appendix for Chapter 2

A.1 Model Development

A.1.1 Model Selection Strategy

A regression model that captures the unobserved area-specific (e.g., difference in biophysical factors, forest size, climate, and management style) and time-specific characteristics (e.g., prices and meteorological event affecting all areas) was used to link the production of different roundwood types with the observed area of disturbed forest and its associated disturbance intensities. The equation of the model can be written as:

$$Y_{it} = \alpha_i + \beta X_{it} + \tau_t + \varepsilon_{it} \quad (\text{A.1})$$

where Y_{it} , in this specific case, is the roundwood production survey data by species and by geographic location (e.g., county), i indexes unit-areas and t the time period. X_{it} denotes the area of forest disturbance in each of the forest type (such as hardwood or softwood or mixed) available and the degree of disturbance intensity in each unit area of the analysis. The index of disturbance intensity is divided into different classes based on its values, ranging from slight thinning to complete clear-

cutting. α_i denote the unit-areas effects, β is the vector of partial effects, τ_t the time specific effects, and ε_{it} the standard random error term. If the α s and τ s are treated as fixed, the model can be estimated by Ordinary Least Squares (OLS) using dummy variables (see Wooldridge, 2010 ([Wooldridge, 2010](#)), for example, for more details on Panel Data Models). It is assumed that the disturbance areas in each type of forest only contribute to the production roundwood of the corresponding species type and disturbance areas in mixed wood forest contribute to that of both species types. The observed disturbance area for each forest type in each unit area is further partitioned according to the disturbance intensity of the pixel which can distinguish between clear-cutting and different degrees of harvesting. The larger the forest disturbance index value, the more severe the forest disturbance ([Huang et al., 2009](#)).

Four models have been tested to estimate roundwood production from the area of forest disturbance with magnitude. The aim is to compare different models in terms of consistency, efficiency, and unbiasedness in order to choose the correct form of the model to estimate the production of different timber types. The four models used to estimate [A.1](#) (for details see, e.g., [Wooldridge, 2010](#)):

1. The pooled Ordinary Least Square (OLS) model
2. The Fixed Effect model
3. The Random Effect model
4. The Two-way Fixed Effect Model

The following tests were used to determine the preferred form of the model based on the guidelines from (Hsiao, 1986) and (Verbeek, 2008). Each test is performed in all six types of roundwood and the two total roundwood types.

1. Test for heterogeneity

The F Test was used to test the potential unobserved heterogeneity in the unit areas. In R, the `pFtest` function in the `plm` package was used to perform this test (Croissant & Millo, 2008). Results of the test on all models shows that all the p-values are less than 0.01, which suggest to reject the null hypothesis.

2. Unit Root Test

The Augmented Dickey-Fuller (ADF) Unit Root Test was used to test whether the variables used are stationary over time. The null hypothesis of the test is that the series is not stationary (has a unit root) over time. The `adf.test` function in the `tseries` package in R (Trapletti & Hornik, 2007) was used to perform the test. Results of the test on all models shows that the p-values are less than 0.01, which suggest to reject the null hypothesis. Since the series has no unit root, the within estimator can be used to estimate the model.

3. Hausman Test

The Hausman Test was used to test whether the Random Effect or the Fixed Effect Model should be used. The `phtest` function in the `plm` package does this test (Croissant & Millo, 2008). Results of the test on all models shows that the p-values are all less than 0.01, which suggest to reject the null hypothesis.

The fixed effect model should be used.

4. Test for Time Effect

The F test was used to test the significance of the time effect (Greene, 2007, chap. 9). In R, the `pFtest` function in the `plm` package does this test (Croissant & Millo, 2008). Results of the test on all models shows that the p-values are all less than 0.01, which suggest to reject the null hypothesis. Time effect should be taken into account.

A.1.2 Industrial Roundwood Production Unit Conversion

All the units of the industrial roundwood production are converted to cubic meter. In the website that contains the TPO reports and the other two publications that contain TPO data of 1992 and 1994, the unit of reporting is in thousand cubic feet. Conversion factor of 1 cubic meter equals to 35.3147 cubic feet is used to convert thousand cubic feet to thousand cubic meter. On the other hand, for the annual pulpwood production records in the 80s, the unit of reporting is in standard cords. The conversion factor of softwood pulpwood is 72.5 cubic feet per cord and that of hardwood pulpwood is 76.6 cubic feet per cord in North Carolina, according to the reports of the North Carolina assessment of TPO and use (Cooper & Mann, 2009; Cooper et al., 2011; Howell & Brown, 2004; Johnson & Brown, 1999; Johnson, 1994; Johnson & Brown, 1996, 2002; Johnson et al., 1997). The unit for the annual pulpwood production records in the 80s was converted to cubic feet and

then converted to cubic meter.

A.1.3 Dependent and Independent Variables

The dependent variables used in the model of this study are the survey of the volume of each type of timber (roundwood) produced in each county, called the Timber Product Output (TPO) record. Data are available for eight years: 1995, 1997, 1999, 2001, 2003, 2005, 2007, and 2009. Data of additional two years, 1992 and 1994, are available in two assessment reports in print ([Johnson, 1994](#); [Johnson & Brown, 1996](#)). Sawlog and pulpwood, of both softwood and hardwood types, consist of about 85% of roundwood production in North Carolina. Although fuelwood is considered as a by-product for timberland management ([Cooper & Mann, 2009](#)), it consists about 10% of the total industrial roundwood production ([Cooper & Mann, 2009](#); [Cooper et al., 2011](#); [Howell & Brown, 2004](#); [Johnson & Brown, 1999, 1996, 2002](#); [Johnson & Steppleton, 1997](#)). Other miscellaneous products, such as chips, post, poles, and pilings consist of about 1% of the total roundwood produced ([Cooper & Mann, 2009](#); [Cooper et al., 2011](#); [Howell & Brown, 2004](#); [Johnson & Brown, 1999, 1996, 2002](#); [Johnson et al., 1997](#)). Because the miscellaneous products consist of such small percentage of the roundwood produced and do not fit in either the sawlog, pulpwood, or fuelwood type, their annual volume production are not estimated in this study. Additional pulpwood data are available every year in the Southern Pulpwood production reports in print ([Howell, 1993](#); [Howell & Hartsell, 1995](#); [Hutchins Jr., 1991](#); [Hutchins Jr., 1989](#); [Johnson & Steppleton, 1997, 2000](#),

2004, 2006, 2008; Johnson et al., 2010; May, 1988; Vissage, 1991; Vissage & Miller, 1992).

The independent variables used in the model of this study are the forest disturbed area per county and its associated disturbance intensity. The disturbance intensity information is obtained from the Forest Index of the VCT product (Huang et al., 2010) (henceforth, “Magnitude”). It indicates the severity of the forest disturbance. The values of the forest disturbance magnitude range from 1 to 255. The higher the value, the more intense the disturbance. Plots of the distribution of the annual disturbance magnitude are used to analyze how to categorized the magnitude data. Based on the change in slope and the general understanding of the nature of the data, the disturbance magnitude are divided into 4 categories based on the range of the values: a. less than 30; b. 30–60; c. 61–90; d. greater than 91. The impact of the disturbance events with magnitude values in category a are the least severe, which is likely to be thinning; that with magnitude values in category d are the most severe, which is likely to be clear-cutting. Categories b and c have the disturbance severity somewhere in between, which could be selective logging. A given disturbance area in a county is petitioned into the above four categories stated. It is assumed that the disturbance occurred in the same type of forest would yield the same type of industrial roundwood. The next section provides details of how to use the disturbed mixed wood forest area as a variable in the model to estimate the production of both hardwood and softwood roundwood.

The definition of a recording year is different for the TPO data and the forest

disturbance events. Forest disturbance data are obtained from the satellite remote sensing product, the Vegetation Change Tracker (VCT) , which captures the disturbance events occur some time during the growing season (May to September) to the next growing season—Disturbance events happen in the months after the growing season in the end of a calendar year will be counted as disturbance in the following year. However, TPO data record timber production based on calendar years. In order to reconcile the time difference between the forest disturbance data and the TPO data, two years of forest disturbance data are used as the independent variables to estimate roundwood production. The next section provides details of how to use the time difference in the forest disturbance area as a variable in the model to best estimate the production of both hardwood and softwood roundwood.

A.1.4 Possible Interactions Effects

Three models were used to account for both, the impact of disturbance on mixed wood forest, and the timing of the disturbance events by varying the combination of the independent variables. In the first model, the independent variables are the disturbance areas in the mixed wood forest and the disturbance areas in both softwood and hardwood forests. No interaction terms were included in the models. For the second model, the main purpose is to test the interaction between the timing of harvest. Lastly, the third model is to test whether there is any interaction between hardwood or softwood forest disturbance with that in the mixed wood forest.

Let Y be the roundwood production; XE be the disturbance area in earlier half of the year of either the hardwood or the softwood forest and ME be that of the mixwood forest; XL be the disturbance area in earlier half of the year of either the hardwood or the softwood forest and ML be that of the mixwood forest; α_i are dummy variables for each county; subscripts i and j indicate the category of the magnitude which the forest disturbance belongs to, where j and k are integers range between 1 and 4.

1. The equation for the model with no interaction looks as follow:

$$Y = \alpha + \sum_j \beta_j (XE + ME)_j + \sum_j \delta_j (XL + ML)_j \quad (\text{A.2})$$

2. The equation for the model with possible interactions between disturbances in different time period looks as follow:

$$Y = \alpha + \sum_j \beta_j (XE + ME)_j + \sum_j \delta_j (XL + ML)_j + \sum_j \sum_k \gamma_{jk} (XE + ME)_j (XL + ML)_k \quad (\text{A.3})$$

The hypothesis is that the forest harvest amount in earlier months of the year may affect that of the latter months of the year.

3. The equation for the model with possible interactions with mixwood forests

looks as follow:

$$\begin{aligned}
Y = & \alpha + \sum_j \beta_j X E_j + \sum_j \delta_j M E_j + \sum_j \gamma_j X L_j + \zeta_j M L_j \\
& + \sum_j \sum_k \eta_{jk} X E_j M E_k + \sum_j \sum_k \xi_{jk} X L_j M L_k
\end{aligned} \tag{A.4}$$

The hypothesis is that the disturbance events occur in the softwood or hardwood forests may influence the amount of harvest in the mixed wood forests.

A.1.5 Model Comparison Result

To select the best functional form for the model, the second-order corrected Akaike Information Criterion (AIC), the regression root mean square errors (RMSE), and the ten-fold cross validation RMSE were used. AIC is a goodness of fit measure. Model with lower AIC is preferred (Verbeek, 2008, chap. 3). The second-order corrected AIC (henceforth, AICc) is the extension of the AIC that takes into account the number of parameters in relation to the number of observation. When the number of observation is large enough, the second-order corrected AIC converges to AIC. The AICc function in the qcpR package in R (Ritz & Spiess, 2008) was used to calculate the second-order corrected AIC value for each model. The RMSE values for the robust regression result are calculated using the RMSE function in the qcpR package in R (Ritz & Spiess, 2008). The regression RMSE values shows the difference between the estimation value and the survey values. The lower the regression RMSE value, the better the model. The major problem of increasing

the number of independent variables is, although it increases the goodness of fit, it tends to cause over-fitting. The best functional form of the model should not only increase the goodness of fit, but also not cause the over-fitting problem.

To test the stability and the validity of the models, ten-fold cross validation and leave-one-year out cross validation were performed. For the ten-fold cross validation, 90% of the data are used as training while 10% of the data are used for testing. This is used to prevent over-fitting. The average RMSE of each model was reported. The leave-one-year out cross validation was used to test if the model can be used to estimate the roundwood production when the survey data do not exist. Not only a smaller RMSE value is preferred for both validations, but also the values should be close to that of the regression analysis if the model is consistent. In addition to the two cross validation analyses, the more complete pulpwood data was also used to compare the validity of the models in the years when the data is missing. County-level pulpwood data are available for every year in the southeastern US. The estimation results for both hardwood and softwood pulpwood by each model are tested against the annual data provided by the US Forest Service. The RMSE values of hardwood pulpwood and softwood pulpwood of each model are reported, in addition to the RMSE of both types of pulpwood in each ecoregion.

Although the AICc values for the model that has no interaction between independent variables are the highest for most types of roundwood, that may be caused by over-fitting. From different types of RMSE values, independent variables that consist of interaction terms between the disturbances in softwood and hardwood

forest with mixwood forest is the best form of the model. It has the lowest RMSE values for different tests for most of the roundwood product type. Although for hardwood pulpwood, it does not have the lowest RMSE values for all the tests, especially when the estimation is split according to forest survey regions, it has the lowest RMSE for most of other wood types.

A good sign that all the models are consistent and can be used to estimate roundwood production for the years when the TPO data do not exist is that the regression RMSE values are very similar to that in the leave-one-year-out test and in the 10-fold cross validation.

A.1.6 Model Result for Roundwood Production

The first step in getting the annual carbon influx to the HWP is to obtain the annual roundwood production at the county level. Figure A.1 shows the scatter-plots of survey data plotted against the model’s predicted values for each type of roundwood product. The estimates for all the are combined in one plot. The plots show that most of the points for each type of the roundwood are scattered symmetrically about a 45 degree line. For hardwood pulpwood (HP), the plot shows that the model cannot make a good prediction for some of the more extreme values of the HP produced.

A.2 Uncertainty and Error Propagation

When calculating the state-level estimation confidence interval, the rules of error propagation have to be applied. The calculation follows the rules stated in Chapter 3 in *An Introduction to Error Analysis: The Study of Uncertainties in Physical Measurements* (Taylor, 1996). The general formula for error propagation is as follow:

For measurements x_1, x_2, \dots, x_n with uncertainties $\delta x_1, \delta x_2, \dots, \delta x_n$, q is a function of x_1, x_2, \dots, x_n , where $q = f(x_1, x_2, \dots, x_n)$. The the uncertainty in q is:

$$\delta q = \sqrt{\left(\frac{\partial q}{\partial x_1} \delta x_1\right)^2 + \dots + \left(\frac{\partial q}{\partial x_n} \delta x_n\right)^2} \quad (\text{A.5})$$

In the case of this study, each county has an estimate of roundwood production

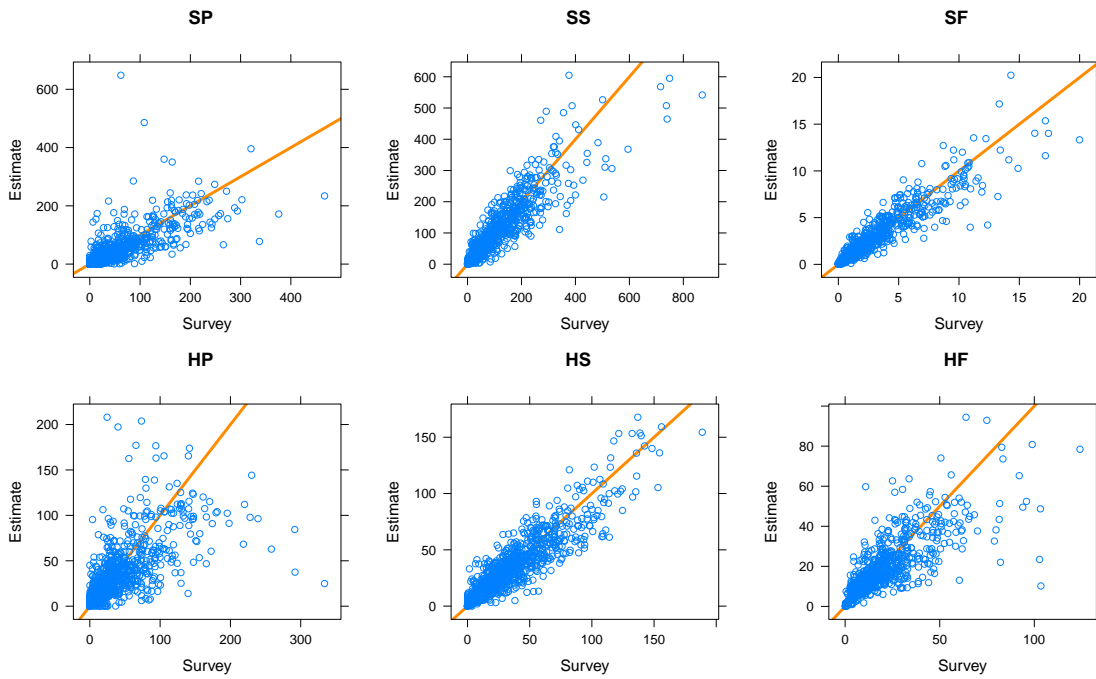


Figure A.1: Scatterplots of county survey data plotted against the model's predicted values (in thousand cubic meter) for each type of roundwood product and for all available years. Estimation values against the TPO survey value for each type of roundwood production (in thousand cubic meter). The orange-colored line represents the one-to-one line.

x_n and a related uncertainty, δx_n , where n is an index indicating the n^{th} county and uncertainty is the margin of error, which equals to the difference between the upper limit of the 95% confidence interval and the best estimate of roundwood production. Because the variables used in the model are log transformed, the state-level roundwood production, q , is calculated as follow:

$$q = e^{x_1} + e^{x_2} + \dots + e^{x_n}, \quad (\text{A.6})$$

where $\frac{\partial q}{\partial x_n} = e^{x_n}$. Then, the uncertainty of q is:

$$\delta q = \sqrt{(e^{x_1} \delta x_1)^2 + \dots + (e^{x_n} \delta x_n)^2} \quad (\text{A.7})$$

For the annual roundwood production (t) that consists of the sum of the annual production of softwood products (a) and hardwood products (b), its margin of error is calculated as follow:

$$t = \text{SoftSG} * (a_1 + a_2 + a_3) + \text{HardSG} * (b_1 + b_2 + b_3) \quad (\text{A.8})$$

$$\delta t = \sqrt{(\text{SoftSG} * (a_1 + a_2 + a_3))^2 + (\text{HardSG} * (b_1 + b_2 + b_3))^2} \quad (\text{A.9})$$

where SoftSG and HardSG are the specific gravity of softwood and hardwood respectively.

A.2.1 Error Propagation in Linear Trend

To test if there is any trend in the total roundwood production over the years, the weighted least square fit is used to fit the time series of the total roundwood

production. The weighted least square fit is the generalization of the least square fit where inverse of the uncertainty of each estimation is the weight, w , and is used to calculate the parameters and the uncertainty of the fit. If there is no uncertainty in the measurement, the equation of the weighted least square fit converges to least square fit (Taylor, 1996, chap. 8). Let the equation of the fit line be $y = mx + c$ and the weight of the i th measurement as $w = 1/(\sigma_i)^2$, where σ_i is the standard deviation of the i th measurement; the calculation of the slope, m , and its uncertainty, σ_m , is as following:

$$m = \frac{\sum wx^2 \sum wy - \sum wx \sum wxy}{\Delta}, \quad (\text{A.10})$$

where $\Delta = \sum w \sum wx^2 - (\sum wx)^2$

$$\sigma_m = \sqrt{\frac{\sum wx^2}{\Delta}} \quad (\text{A.11})$$

If the absolute value of the uncertainty of the slope of the trend line is greater than the absolute value of the slope of the trend line, then any trend calculated for the total roundwood production is insignificant.

A.2.2 Contribution From the Input Data

Contribution to the estimation errors also comes from the input data that were used in the model. Huang et. al (2010) (Huang et al., 2010) summarizes three types of contribution to the error of the disturbance maps: it is difficult to detect non-standing clearing, such as thinning, by the current version of the VCT algorithm; some crop fields and wetlands are misclassified as forest; marginal forests in semiarid

environment are difficult to distinguish from other cover types. Misclassification was minimized by using an independent forest classification map (McKerrow et al., 2006) to augment VCT. While this map also has errors, the combination was found to reduce crop and wetland errors. Errors in semiarid forests are not applicable to NC. Advances in detection technology and methods of analysis can be expected to improve the VCT algorithm and could reduce the impact of the errors in the future.

The timber survey data (TPO) data used to calibrate the model in this study are based on reports from the wood processing mills. However, the uncertainty associated with these data is not known. While the “miscellaneous” roundwood class in the TPO report was not considered in this study, its share of the total roundwood production was negligible. Although it is not likely to have affected the results described here, the omission of this class could affect the method when applied elsewhere.

A.3 Calculation of NC’s Roundwood Production Share of the Continental US

In order to calculate the percentage of the roundwood production of NC out of that for the continental United States, the roundwood output data of NC and the continental United States from the Resources Planning Act (RPA) Assessment reports were used. The RPA Assessment reports were available for years 1997, 2002, and 2007. Data for the 2010 report will not be used. The data for the production

of NC are obtained from the Core Table 2—Volume of industrial timber harvested by timber product, major species group and year and the data for the production of the continental United States are obtained for the "Total removals" column of the RPA fact sheet. All of those data can be obtained at the TPO website([U.S. Department of Agriculture, 2015](#)). Results of the percentage calculation are shown in table [A.2](#).

A.4 Conversion of Wood Volume to Wood Mass

Volume of roundwood product can be converted to mass by using the green specific gravity (SG_{gr}). SG_{gr} is density of wood divided by the density of water (ρ_w) based on wood dry mass associated with green (freshly cut) tree volume, so it is unit-less [Miles & Smith \(2009\)](#). Density of water is $1 \text{ g/cm}^3 = 1 \text{ t/m}^3$. Therefore, mass of dry wood (M), in tons, can be obtained as follows:

$$M = \text{Volume} \cdot SG_{gr} \cdot \rho_{H_2O}, \quad (\text{A.12})$$

A.5 Comparison with Other Estimation Approaches

There are two other estimation approaches that can give the same level of detail as the method used in this paper, namely, *linear interpolation* of the TPO survey data and *conversion factor* of forest disturbance area to volume of reoundwood produced. However, their estimation results are either in shorter length of time period or lack the details of different roundwood type.

A.5.1 Comparison with Linear Interpolation

The linear interpolation method cannot estimate the roundwood production beyond the range of years of the survey data. Since the roundwood data are only available after 1992, linear interpolation can only obtain the production estimate for the missing years between 1992 and 2009. To extend the estimate beyond the available survey years, linear extrapolation of the data has to be used in combination with linear interpolating the data. Because there are some increasing trends in some of the roundwood in some counties, if linear extrapolation/extrapolation method used in Adams et. al (2006) ([Adams et al., 2006](#)) was followed to obtain estimation of the roundwood production in the 80s, then some of the estimations would be negative.

Experiments were performed to compare the proposed method with the linear interpolation/extrapolation method in terms of estimating the past and the future roundwood production based on limited years of TPO data. For both the estimation in this study and the linear interpolation/ extrapolation methods, the first 3 available TPO years were used to estimate each of the last 3 years of TPO data, and the last 3 available TPO years data were also used to estimate each of the 3 earliest TPO data. For most of the states that are not in the Southern region, the TPO data availability can be as limited as to three years before 2009 ([U.S. Department of Agriculture, 2015](#)). The robust trimmed mean squared error approach was used to calculate the root mean square error (RMSE) of each estimation years with the

TPO values. Tables [A.3](#) and [A.4](#) show the results of the RMSE of the comparison. It shows that estimation results using the proposed method in this study is significantly better than those using the linear interpolation/extrapolation method when limited number of years of survey data is available.

Table A.1: The first and second sections of the table (this page) show the root mean square errors (RMSE) of the tests described above and the AIC values of each model; the third section (next page) shows the RMSE for the additional pulpwood data in each ecoregion. The Add Pulp columns (this page) show the RMSE using the additional pulpwood data as testing. All the RMSE values are based on log-transformed models. For the wood types, HP=Hardwood pulpwood, HS=Hardwood sawlog; HF=Hardwood Fuelwood; SP=Softwood pulpwood, SS=Softwood sawlog; SF=Softwood Fuelwood. For the forest survey regions: NCP=Northern Coastal Plain; SCP=Southern Coastal Plain. Numbers with * indicate the lowest value among models.

Wood Type	Test Type	No Int.	Add Pulp	Time Int.	Add Pulp	Mix Int.	Add Pulp	
HP	Reg.	1.94	2.04	1.84*	1.97*	1.88	2.00	
HS		0.84		0.82		0.82*		
HF		0.55		0.53		0.52*		
SP		2.40		2.37		2.22*		2.31
SS		0.67		0.65*		0.65		
SF		0.66		0.64		0.64*		
HP		10-Fd. CV		1.70				1.60*
HS	0.84		0.83	0.82*				
HF	0.59		0.56	0.56*				
SP	2.34		2.31	2.16*				
SS	0.60		0.58	0.58*				
SF	0.71		0.69	0.69*				
HP	Time CV		1.83		1.74*			1.78
HS		0.76	0.75		0.75*			
HF		0.54	0.52		0.51*			
SP		2.15	2.11		2.08*			
SS		0.63	0.62*		0.62			
SF		0.64	0.63		0.62*			
HP		AICc	4403*			4339		
HS	2730*		2737	2792				
HF	1565		1547*	1588				
SP	4833*		4850	4780				
SS	2267		2247*	2315				
SF	1846*		1862	1923				

Table A.1 (cont'd)

Wood Type	Test Type	No Int.	Add Pulp	Time Int.	Add Pulp	Mix Int.	Add Pulp
Survey Region							
HP	NCP		1.05		1.12		1.08*
HP	SCP		1.30		1.29*		1.30
HP	Piedmont		1.70		1.70		1.68*
HP	Mountain		3.58		3.36*		3.49
SP	NCP		0.91*		0.93		0.93
SP	SCP		1.42*		1.43		1.46
SP	Piedmont		2.30		2.29		2.29*
SP	Mountain		3.87		3.79*		3.86

Table A.2: Proportion of the roundwood produced in North Carolina to that of the continental United States in each RPA report

Year	NC	US	Percentage
1997	9.54E+05	2.12E+07	4.50
2002	8.53E+05	2.02E+07	4.22
2007	8.59E+05	2.12E+07	4.05

Table A.3: RMSE in thousand cubic meter of using TPO years 2005, 2007, and 2009 to backward project the first 3 available TPO years (1992, 1994, and 1995) using Linear Interpolation/Extrapolation method and the estimation method in this study. The Forest Service divides North Carolina into four regions: Northern Coastal Plain (NCP), Southern Coastal Plain (SCP), Piedmont (Pied), and Mountain (Mont). Types of roundwood are: HS=Hardwood Sawlog; HP=Hardwood Pulpwood; HF=Hardwood Fuelwood; SS=Softwood Sawlog; SP=Softwood Pulpwood; SF=Softwood Fuelwood.

Year	Linear Interpolation/Extrapolation					Estimates in this study					
	NCP	SCP	Pied	Mont	Total	NCP	SCP	Pied	Mont	Total	
1992											
	HP	140.0	107.1	85.5	81.8	105.0	38.4	42.1	21.8	23.4	32.5
	HS	69.5	67.7	80.9	107.9	80.4	21.3	30.0	45.5	22.6	25.3
	SP	35.0	58.3	29.6	29.3	39.3	122.1	90.4	145.1	9.8	103.1
	SS	192.3	188.0	122.5	64.1	151.0	127.4	131.2	65.5	41.1	99.0
1994											
	HP	116.0	102.4	76.4	74.2	93.4	38.2	49.3	26.8	32.9	37.7
	HS	57.5	57.7	69.9	87.2	67.7	24.7	27.2	21.9	21.5	24.2
	SP	121.3	87.3	55.4	44.1	77.2	44.3	53.6	54.4	33.8	47.9
	SS	190.6	192.4	96.7	66.7	146.5	64.4	52.1	70.6	15.1	56.4
1995											
	HP	105.6	94.5	68.1	73.4	86.0	37.2	48.3	32.1	28.7	37.8
	HS	51.0	54.9	65.8	79.7	62.8	24.2	30.5	22.7	17.1	24.8
	SP	100.0	85.4	53.8	29.1	69.1	45.6	58.8	62.5	10.5	50.1
	SS	157.2	166.7	85.9	58.9	124.9	93.6	68.4	64.1	33.0	67.7

Table A.4: RMSE in thousand cubic meter of using TPO years 1992, 1994, and 1995 to forward project the most recent 3 available TPO years (2005, 2007, and 2009) using Linear Interpolation/Extrapolation method and the estimation method proposed in this study. The Forest Service divides North Carolina into four regions: Northern Coastal Plain (NCP), Southern Coastal Plain (SCP), Piedmont (Pied), and Mountain (Mont). Types of roundwood are: HS=Hardwood Sawlog; HP=Hardwood Pulpwood; HF=Hardwood Fuelwood; SS=Softwood Sawlog; SP=Softwood Pulpwood; SF=Softwood Fuelwood.

Year	Linear Interpolation/Extrapolation					Estimates in this study				
	NCP	SCP	Pied	Mont	Total	NCP	SCP	Pied	Mont	Total
2009										
HP	95.4	105.4	75.7	52.4	85.9	17.8	18.2	28.0	24.9	23.1
HS	71.8	65.0	57.7	77.7	67.3	18.3	30.4	35.7	16.6	27.6
SP	266.6	361.6	197.2	103.0	253.9	128.9	88.1	40.0	12.4	78.8
SS	389.3	414.6	245.3	185.6	322.6	295.1	267.5	60.6	44.8	197.3
2007										
HP	84.3	94.7	67.4	45.0	76.4	26.5	30.9	31.0	32.1	30.3
HS	59.3	54.4	49.8	61.5	55.7	15.1	28.1	21.2	19.0	22.1
SP	247.5	314.9	178.3	88.8	225.8	90.4	80.1	13.9	13.8	55.5
SS	360.0	379.4	213.6	160.9	292.6	126.0	142.9	47.8	36.9	99.7
2005										
HP	71.5	128.6	90.1	54.9	94.3	42.3	83.0	36.6	48.6	56.9
HS	37.7	48.9	43.8	57.9	47.3	16.4	28.3	18.3	24.5	22.6
SP	228.2	282.3	148.3	73.8	200.5	62.4	45.4	20.8	20.3	38.6
SS	294.3	331.8	183.6	143.6	249.5	103.4	105.5	40.0	21.0	76.8

A.5.2 Comparison with approaches using a Conversion Factor

Although using the conversion factor approach can estimate roundwood production as far back as when the forest disturbance area information exist, it cannot provide the detail information of the types of roundwood produced. Conversion factor for the southeastern United States calculated by Masek et. al (2011) [Masek et al. \(2011\)](#) is $100 \text{ m}^3/\text{ha}$, which is used to estimate the total amount of annual roundwood production. The conversion factor is not only a very general estimation for regions in North America, but also derived from the total roundwood production, which includes both softwood and hardwood. Thus, it cannot be used to estimate different types of roundwood. The range of the dates of forest disturbance areas calculated by VCT starts from a day in the growing season (May–September) and ends at a day in the next growing season. Any forest disturbance event happens after the end of the growing season would be captured as the disturbance event for the following year. The comparison for carbon in HWP with other estimation method is based on calendar year. In order to reconcile the timing difference between the VCT data and the calendar year, for the estimation using conversion factor, the annual forest disturbance area in a calendar year is calculated by taking the average of the two years of the forest disturbance areas. In addition, since the estimation from the conversion factor method is for both hardwood and softwood, when converting the volume of the total roundwood production to carbon mass, the average value of the specific gravity of hardwood and softwood was used.

Table A.5: Comparison of the average C in roundwood (in Mt C/year) between the proposed estimation method in this study and others: CF=Conversion Factor; LI=Linear Interpolation.

C in Roundwood by Ecoregion (Mt C/yr)					
	NCP	SCP	Piedmont	Mountain	Total
CF	-1.36	-1.81	-0.72	-0.38	-4.27
LI	0.05	0.14	0.08	0.06	0.33

A.6 Analysis of the Impact of Hurricanes

Besides harvest, the most common forest disturbance in North Carolina is related to wind storms . For the period in this study study, NC was hit by hurricanes in most years¹. For the years when TPO data is available, only 1992, 1997, 2001, and 2007 have not experienced significant hurricane related disruptive events. In 1999, two major hurricanes have made landfall. More detailed cross-validation results were added to assesses the model’s predictive accuracy for years with reported hurricanes (see, Table A.6 and SI). The mean square error (MSE) of the model estimates were compared with the corresponding TPO data for each year used as testing while the rest of the years as training dataset.

The similarity in the MSE between years shows that the hurricanes did not significantly affect the relation between forest disturbance areas and roundwood production overall during the study period (see Figure A.2). A factorial analysis

¹See, e.g., <http://nc-climate.ncsu.edu/climate/hurricanes/landfalling.php> or <https://www.nccrimecontrol.org/Index2.cfm?a=000003,000010,000025,000185,001329>.

Table A.6: Mean Square Error (MSE, in square of 10^3 cubic meter) results of cross validation for roundwood production estimates by reserving one year as testing for all roundwood estimates. Production units are thousand cubic meter. Years with no significant hurricane landfalls are marked in GreenYellow.

Year	Softwood			Hardwood		
	Pulp	Sawlog	Fuel	Pulp	Sawlog	Fuel
1992	4.27	0.80		1.24	0.58	
1994	2.21	0.57		1.44	0.43	
1995	2.07	0.40	0.39	1.44	0.45	0.54
1997	2.07	0.43	0.65	1.30	0.78	0.53
1999	1.97	0.46	0.46	1.13	0.31	0.44
2001	1.74	0.45	0.68	2.67	0.66	0.44
2003	1.50	0.80	0.70	2.03	0.92	0.47
2005	1.41	0.59	0.50	2.96	0.82	0.78
2007	1.84	0.71	0.68	2.04	0.94	0.42
2009	1.72	0.97	0.90	1.55	1.57	0.43

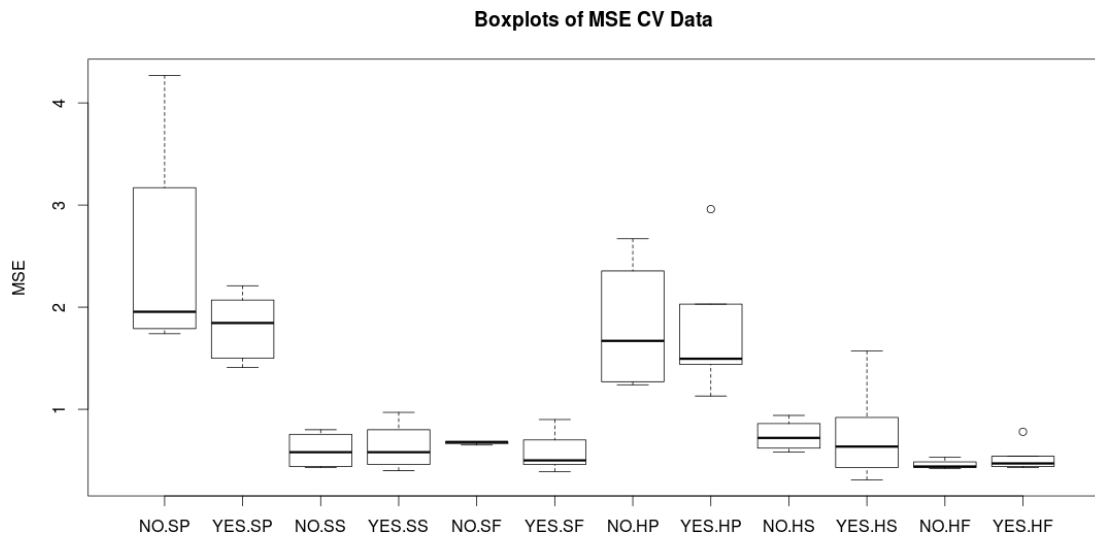


Figure A.2: Boxplot of MSE CV results. YES means year with a hurricane, NO otherwise

Table A.7: Factorial analysis of the MSE between years with and without hurricane damage

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
wood type	5	22.47	4.49	19.48	0.0000
hurricane	1	0.20	0.20	0.88	0.3536
wood type:hurricane	5	0.89	0.18	0.78	0.5723
Residuals	44	10.15	0.23		

confirms that the variability of MSE between years with and without hurricane damage, is not significantly different from within variability, therefore no evidence showed that hurricanes adversely affect the predictive power of the model.

For coastal counties that are frequently impacted by hurricanes, the county-specific dummy variable (*alpha*) in the model would have taken that into account. It is important to note that for forests affected by wind damage, about 15% of the fallen trees will be harvested immediately after the events as salvage logging (McNulty, 2002). This management practice mitigates the contribution of wind damage to roundwood harvest. In the proposed framework, if more detailed and specific information on natural disturbance events is available, the prediction accuracy can improve substantially.

Appendix B: Appendix for Chapter 3

B.1 Land Types

To simulate the interactions between plant communities, species with unique attributes are grouped into ecoregions, where the ecological and geological characters are similar. The fine resolution of ecological zone of Western North Carolina of [Simon et al. \(2005\)](#) was used. Eleven ecological zones were classified for the Southern Appalachian upland forest community, in addition to one ecological zone that is not classified. In addition to ecozone, Landis-ii also requires initial community files that define the species present, its location, and its age. The initial community map is the forest type defined by the North Carolina GAP Analysis project ([McKerrow et al., 2006](#)), which further classifies forest type based on the National Land Cover Data (NLCD) 2001 data by using local forest maps and survey.

B.2 Initial Condition

Age values for individual species that are present in the study area. Individual tree species fall into the different initial community types in the study area of Western North Carolina were obtained from the data from 2002 to 2010 in the

forest inventory and analysis (FIA) database ([U.S. Department of Agriculture, Forest Service, 2016](#)). Although the FIA does not provide the exact location of the measurement sites, the data can be used as samples of the forest type where a site belongs to and the FIA values can be applied to the whole of that type. The age for the tree species that has the breast height diameter of less than 3 inches is considered to be aged 10. For those trees that FIA does not provide age information, the growth index curve from [Carmean et al. \(1989\)](#) for each species is used to estimate its age. The Site Index Curve equation requires height, site index, and species-specific parameters to obtain the age value of specific tree. The species-specific parameters were obtained from the Southern Variant Overview Forest Vegetation Simulator ([Keyser, 2008](#)). For the species-specific parameters that are not available for some species, the values of those of the species from the same genus group that have similar physiology are used as substitution. For those species that the FIA database does not provide the height or the site index value, the average value of the same species in the same initial community type will be used. Age estimations will be grouped into age class cohort that is equal to the succession time step, which is 10 years. For example, trees of age 1-10 will be referred to as age 10. 36 tree species were present in the study area between 2002 and 2010 according to the plot data from the FIA. Twenty species are used in the simulation. They are selected based on abundance.

B.3 Climate Data

No climate change scenario will be simulated in this study. The monthly temperature and precipitation input data for each ecoregion are the monthly average data from 1980 to 2010 acquired from the PRISM climate data website at resolution of 0.0417 degree ([PRISM Climate Group, Oregon State University, 2004](#)). The standard deviations from the mean of both the temperature and precipitation data for each ecoregion were calculated as part of the input data.

B.4 Species Parameters

One of the required species parameter is leaf longevity. The information of leaf longevity for five of the species in this study are available from the study conducted in the Coweeta Hydrological Laboratory, North Carolina by Hwang et. al (2014) ([Hwang et al., 2014](#)). The five species are *Nyssa sylvatica* (Blackgum), *Acer rubrum* (Red Maple), *Oxydendrum arboreum* (Sourwood), *Quercus prinus* (Chestnut Oak), *Quercus rubra* (Northern Red Oak). The leaf longevity information is obtained by averaging the annual difference between the spring budding date and the fall leaf-off date of each branch. For other deciduous trees that the information is not available, the leaf longevity values are assumed to be the average of that of the closest species. For other evergreen species, the leaf longevity is obtained from the gymnosperm database ([Earle, 2016](#)).

The age-related mortality parameters of the oak trees are estimated based

on the study by [Greenberg et al. \(2011\)](#). The study finds that the red oak group (Northern red oak, Scarlet oak, Black oak) that dies of declined-related cause is on average, 108 years old, and that of the white oak group (White oak, Chestnut oak) is 138 years old. The starting ages of the age-related mortality of other species are determined by when the tree height growth slows down from the growth Site Index Curves in [Carmean et al. \(1989\)](#). For the two pine species (Table Mountain pine and Pitch pine) that do not have the Site Index curve, the average value of the mortality curve shape parameter of the other pine species was used. For the rest of the species that do not have the Site Index Curve available in [Carmean et al. \(1989\)](#), the dbh and height are plotted against age for the Western North Carolina FIA plot. The age when the growth slows down is when the age-related mortality begins.

According to the USDA definition of sawtimber tree, the d.b.h for softwoods and hardwoods must be at least 9 inches and 11 inches respectively (cite). And the minimum d.b.h. for both softwood and hardwood pulpwood is 6 inches (15cm) according to the information in the Woodland Owner Notes ([Bardon, 2002](#)). The age of each species present in the FIA data of Western North Carolina was determined using the site index ([B.2](#)). The minimum age of a species that its dbh reaches around 6 inches (between 5.7 and 6.3 inches) is used as the minimum age for merchantable stems.

B.5 DOM pool

The DOM pools table requires the input of proportion of carbon in the decay material that transferred to the atmosphere for each DOM and soil pool. The values are obtained from [Kurz et al. \(2009\)](#).

B.6 Ecoregion Species DOM Parameters

The base decay rate in each pool and the Q_{10} value for a reference temperature of 10 °C of each species are obtained from [Kurz et al. \(2009\)](#). Russell et. al (2014) ([Russell et al., 2014](#)) provides the decay rate of the down dead wood of diameter greater than or equal to 7.62 cm for the species in the forest of the eastern United States. It corresponds with the "snag stem" pool and the "snag other wood pool". The values provided by this study are used as the "snag other wood pool" base decay rate. The "snag stem" decay rate is approximated as 2.5% more than than of the "snag other wood pool" based on the values reported in [Kurz et al. \(2009\)](#). If the value of the species is not reported in the study, then either the average value of the species in the same genus is used, or the average value of that species wood type. Base decay rate for other pools are obtained from [Kurz et al. \(2009\)](#).

B.7 DisturbFireTransferDOM

No carbon is transferred to the product sector after fire.

B.8 DisturbOtherTransferDOM

15% of the carbon in the snag stem is transferred to the product sector after windthrow events to indicate possible salvage logging ([McNulty, 2002](#)).

B.9 DisturbFireTransferBiomass

No carbon is transferred to the product sector after fire.

B.10 DisturbOtherTransferBiomass

15% of the carbon in the merchantable part of the woody pool is transferred to the product sector after windthrow events to indicate possible salvage logging ([McNulty, 2002](#)).

B.11 MaxBiomassTimeSeries

Values of the maximum biomass are obtained from [Thompson et al. \(2011\)](#). For species that are not mentioned in their paper, their maximum biomass value are assumed to be 250Mg/ha . The unit is converted to g/m^2 in the input file. The value of each species is the same for all ecoregions and periods.

B.12 Root Dynamics Table

The values for root dynamics of yellow poplar are obtained from estimations of the results reported in [Harris et al. \(1980\)](#). The percentage of biomass of the root system of American beech (*Fagus grandifolia*) are obtained from [Santantonio et al. \(1977\)](#). The study also reports that roots comprised 28 % of the forest biomass in oak-hickory and eastern white pine forests ([Harris et al., 1980](#)). The dominant species in those forest are red oak, white pine, black oak, red maple, and scarlet oak. Black cherry, white oak, and hemlock are the sub-canopy species, although found less frequently ([Simon et al., 2005](#)). The average root to shoot ratios under different light treatments of red oak, red maple, black oak, and black cherry of 2-year-old are obtained from [Gottschalk \(1985\)](#). Root to shoot ratio data of chestnut oak, northern red oak, and white oak are also available from [Rebbeck et al. \(2011\)](#) ([Rebbeck et al., 2011](#)). Final root to shoot ratio for Northern red oak is the average of data in [Gottschalk \(1985\)](#) and [Rebbeck et al. \(2011\)](#). Root to shoot ratio of pitch pine is obtained by averaging the values for different treatments and sites in [Greenwood et al. \(2002\)](#). Root to shoot ratios of Table Mountain Pine, Eastern Hemlock, and Virginia Pine are obtained from [Neufeld et al. \(2000\)](#). Root to shoot ratio of flowering dogwood and white oak are available in [Riley Jr \(2001\)](#). Root to shoot ratio of pignut hickory and white oak are available in [Arnold & Struve \(1993\)](#). Final root to shoot ratio for white oak is obtained by averaging the values in [Arnold & Struve \(1993\)](#), [Rebbeck et al. \(2011\)](#), and [Riley Jr \(2001\)](#). Root to shoot ratio

of black locust is obtained from [Mebrahtu & Hanover \(1991\)](#). Root to shoot ratio of blackgum is obtained from [Butterfield et al. \(2004\)](#). The root to shoot ratio for sourwood is estimated by averaging the root to shoot values for all hardwood species in the study. The proportion of the fine wood and coarse wood which dies annually and will be added to the DOM pool are assumed to be 99% and 2% based on the estimation from the results by [Edwards & Harris \(1977\)](#).

B.13 ANPPTimeSeries and Probability of Establishment

Values for aboveground net primary production and the probability of establishment in each ecoregion are obtained from using the estimates from the PnET model for Landis-ii ([Xu et al., 2009](#)).

B.13.0.1 parametrization of the PnET model

Latitude of each ecoregion is obtained by calculating the centroid of the areas that belong to the same ecoregion. The water holding capacity for each ecoregion is obtained by the averaging the values of Root Zone Available Water Storage of all the soil map units located within each ecoregion in the gSSURGO map of North Carolina ([Soil Survey Staff, 2015](#)). The Root Zone Available Water Storage takes into account of the root zone depth of each of the soil component so that its values represent the volume of plant available water that is store of the soil. The monthly CO_2 concentration data is obtained from the Mauna Loa records. The values are

calculated by averaging the corresponding monthly records in between 1980 and 2010. The climate input variables are obtained by averaging of the corresponding variables in all ecoregions. The PAR value is obtained by averaging the PAR values for North Carolina in June and in December available in [Aber & Freuder \(2000\)](#).

B.14 Classification of Commercial and Ecological Importance

Both the commercial rank and the ecological rank, as well as the associated use of each species and the associated role of each species are provided by the forest ranger in the National Forest. The commercial rank is based on the worth of the final product that each harvested species is likely to yield. The ecological rank is based on the role of each species in the forest habitat. For both of the ranks, the lower the index, the more preferable the species.

B.14.1 Calculation of relative commercial price

In order to understand the relative economic values of the species each Commercial Class, the relative commercial prices of hardwood sawlog, softwood salog, hardwood pulpwood, and softwood pulpwood were calculated. Instead of understanding the real commercial value of a species, the relative commercial price was calculated so that the model can prioritize stands that have the highest commercial values when selecting sites for harvesting. Based on the commercial rank of the species, each one of them are assigned to one of the roundwood type that their

harvested products are likely to become. Species that belong to Commercial Class V has a relative commercial price of zero, because all of them are not economically valuable. Relative commercial price was calculated based on the average nominal stumpage price of each types of roundwood product from 1976 to 2014. Price data is available at [Bardon \(2016\)](#). Since the price of sawlog is provided based on thousand board feet (MBF), while that of the pulpwood based on standard cord, the price unit was converted to be based on MBF using the conversion factor provided by Timber Mart South for the southern United States. The conversion factor for softwood is 1 MBF=2.798 standard cord and that for hardwood is 1 MBF=3.02 standard cord ([South, 2007](#)). The nominal price of each type of roundwood product between 1976 and 2014 was averaged to calculate the relative prices. The price of the most expensive roundwood type, softwood sawlog, is normalized to \$100/MBF, and the other types are normalized accordingly. Based on the commercial ranking and the information on the associated use of each species, the most likely end use of the merchantable part of a species was estimated. For species rank that is bigger than 17, they are not likely to provide any economic incentive for foresters to harvest. Each species was assigned a relative price the same as that of its likely end use. The relative price of softwood sawlog is normalized to \$100/MBF. Accordingly, that for softwood pulpwood, hardwood sawlog, and hardwood pulpwood is \$25/MBF, \$88/MBF, and \$22/MBF respectively.

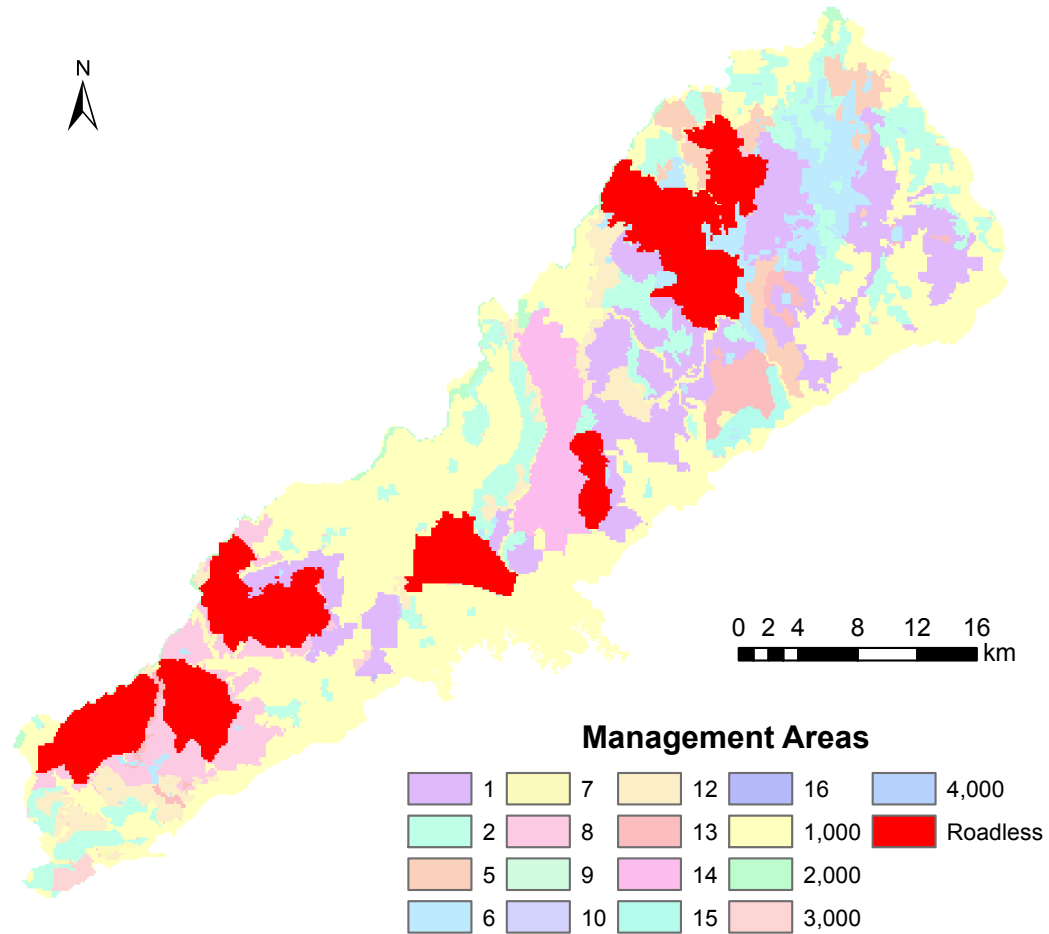


Figure B.1: Management Areas (MAs) designated by the US Forest Service (USFS).

B.15 Harvest Prescription for each Management Area

The USFS divides the District into different management areas (MAs) (fig B.1). In which, areas with MA codes 8, 12, and 14 are considered as ecologically sensitive areas ([National Forests in North Carolina, USFS, 2016](#)).

Each MA has different harvest prescription for both the Aggressive and the Moderate management regimes. The harvest prescriptions in the Aggressive management regime assumes the whole area is profit maximizing by lowering the cost

and maximizing the yield. Both the privately-owned and privately-own forests would employ the clear-cut harvest of different levels of intensity. Ecologically sensitive areas would employ the shelterwoof harvest. In the Moderate management regime, the goal is to maintain sustainable timber harvest while preserving the integrity of the forest landscape. Specifically, publicly-owned forest would employ the management practices similar to that of the USFS, while the privately-owned forest would employ selective harvest of the economically desirable species.

Table B.1: Management practices employed by the USFS in each management area (MA). The MA code corresponds with the MA show in figure B.1. MA codes that are not listed indicate no harvest taken place.

Code	Goal	Aggressive	Moderate
1	Sustainable timber supply	2.5% area/yr Ecological Clearcut	0.48% area/yr Shelterwood harvest and 2% area/yr Group selection
2	Provide a remote forest setting	1% area/yr Ecological Clearcut	0.5% area/yr of cutting understory
5	Scenery with management for timber	1% area/yr Ecological Clearcut	0.36% area/yr Shelterwood harvest and 1% area/yr Group selection
6	Maximum timber yield (privately owned)	2.5% area/yr Economical Clearcut	2.5% area/yr Selective harvest
7	Scenery without management for timber	1% area/yr Ecological Clearcut	0.05% area/yr of Group selection and 0.1% area/yr of Cutting understory
8	Backcountry	0.48% area/yr Shelterwood harvest	0.05% area/yr of cutting understory
10	low activities (Riparian)	1% area/yr of Ecological Clearcut	0.5% area/yr of Cutting understory
12	Black bear habitat (less than 10ha per harvest)	0.48% area/yr Shelterwood harvest	0.24% area/yr Shelterwood harvest and 1% area/yr Group selection

Table B.2: Continue table.

Code	Goal	Aggressive	Moderate
13	Scenery with management for timber	1% area/yr Ecological Clearcut	0.36% area/yr Shelterwood harvest and 1% area/yr Group selection
14	Wilderness area	0.48% area/yr Shelterwood harvest	0.1% area/yr of cutting understory
15	Sustainable timber supply	2.5% area/yr of Ecological Clearcut	0.60% area/yr Shelterwood harvest and 2.5% area/yr Group selection
1000	Maximum timber yield (privately owned)	2.5% area/yr Economical Clearcut	2.5% area/yr Selective harvest
2000	Maximum timber yield (privately owned, with some land protection requirement)	1% area/yr Ecological Clearcut	1% area/yr Selective harvest
3000	Maximum timber yield (privately owned, with strong land protection requirement)	0.48% area/yr Shelterwood harvest	0.1% area/yr of cutting understory and 0.1% area/yr Group selection

For the privately owned areas, if they are not located in the permanently protected areas according to the Land Stewardship data in the NC GAP analysis (McKerrow et al., 2006), then shelterwood harvest and group selection of higher level, just like those MAs designated for sustainable timber supply, were assumed to be practiced in those areas.

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