Quantifying the Carbon Sequestration and Economic Potential of Natural Climate Solutions from Maine's Working Forests

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QUANTIFYING THE CARBON SEQUESTRATION AND ECONOMIC
POTENTIAL OF NATURAL CLIMATE SOLUTIONS FROM MAINE’S
WORKING FORESTS

By
Logan Woodyard
B.S. The Ohio State University, 2020

A THESIS

Submitted in Partial Fulfillment of the
Requirements for the Degree of
Masters of Science
(in Forest Resources)

The Graduate School
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August 2022

Advisory Committee:
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QUANTIFYING THE CARBON SEQUESTRATION AND ECONOMIC POTENTIAL OF NATURAL CLIMATE SOLUTIONS FROM MAINE’S WORKING FORESTS

By Logan Woodyard

Thesis Advisor: Dr. Adam Daigneault

An Abstract of the Thesis Presented in Partial Fulfillment of the Requirements for the Degree of Masters of Science (in Forest Resources) August 2022

The purpose of this thesis is to develop a dynamic, regionally integrated forest sector model framework to identify cost-effective forest management and carbon sequestration practices across Maine’s 17,000,000 acres from 2019 to 2119. We take a three-pronged approach, each with its own set of inputs, parameters, accuracies, and skill level requirements. To achieve this we first review a group of biophysical, spatial, and silvicultural studies in the northeast to determine changes and costs of forest carbon sequestration across different treatments and harvesting practices. This allows us to estimate sequestration above baseline, the cost of mitigation, and determine the strength of Maine’s forest carbon sink over various levels of implementation via landowner participation. Secondly, we analyze forest growth and yield projections (and subsequently carbon sequestration) across 6 different management practices from the Forest Vegetation Simulator (FVS) and determine the net present value (NPV) of each pathway across 100 years. Lastly, we use the commercialized linear programming solver Woodstock to estimate total NPV to the point where harvest levels are maintained simultaneously with increasing forest carbon sequestration under different economic and policy conditions. In Woodstock,
we employ both a regimes-based approach (Model I) and a treatment-based approach (Model II).

Combined, the three approaches create a range of carbon sequestration estimates and total NPV of timber harvests and carbon credits across harvest and area constraints. Both Woodstock models reveal that conducting harvests on stands well-suited to fast growth and recovery can use a mix of intensive, partial harvests, and thinnings can increase forest carbon stocking while also satisfying historical demand for harvested wood products over the long term. This integrated modeling approach will facilitate in developing a more precise forecast over a 100-year horizon, allowing us to determine the carbon price or policy incentive that minimizes the societal cost of increasing mitigation level from Maine’s diverse forest landscape.
ACKNOWLEDGEMENTS

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1 INTRODUCTION

Across the world, forests are highly valued for the vital ecosystem services they provide such as recreation, supporting biodiversity, stabilizing streams and water run-off, carbon sequestration and climate mitigation in addition to providing durable goods like timber, fuel, and fiber. Forests and harvested wood products form the US’s largest net carbon sink, sequestering 775 million metric tons of carbon dioxide equivalent (MMtCO$_2$e/year) from the atmosphere annually or nearly 11% of the nation’s total greenhouse gases (GHGs) emitted (Domke et al., 2021; EPA, 2022). Forest climate mitigation efforts through the creation of "set asides" or reserves in addition to afforestation and reforestation practices, increase the amount of designated forest land which store more carbon than other terrestrial ecosystems (Daigneault et al., 2021; Cook-Patton et al., 2020. In already densely forested regions, many turn to forest management practices to increase tree growth rates, limit tree mortality, and increase forest resiliency thereby sequestering more carbon dioxide from the atmosphere and storing it in biomass and soils (Wade et al., 2022). In addition, manufactured, durable wood products store carbon which comprise around 16% of the carbon sink from forests and harvested wood products nationally (Domke et al., 2021). Together these mitigation efforts and management practices can potentially improve forest health, provide ecosystem service provisions, and enhance carbon sequestration in a time where the negative impacts of climate change are becoming increasingly potent.

Maine has one of the largest forest carbon sinks in the nation, sequestering between 8 and 10 MMtCO$_2$e annually which offsets 60% to 75% of the states reported GHG emissions (Bai et al., 2017; Maine DEP, 2020). Traditionally, Maine has relied upon its 17 million acres of forests to provide both durable and non-durable goods and services (such as recreation and tourism), which comprise nearly 5% of the state’s GDP or $7.7 billion dollars in output to the state economy in 2016 (Bailey et al., 2020). Despite its notable economic contribution, the future of the forest economy is uncertain as technological advances incite drops in consumer demand for traditional pulp and paper, contributing to
the closure of over 8 pulp and paper mills since 2010 (Gan et al., 2009). Over the past 20 years, total statewide harvests declined 28% and the real price of softwood pulp declined 48% (MFS, 2015). Shrinking profit margins for low grade lumber and declining demand for forest products disincentivize improved forest management practices, leaving much of Maine’s forests to naturally regenerate in a potentially unsustainable and uncontrolled trajectory (Gan et al., 2009). Research examining a 20-year change in relative forest density (RD, ranging from 0-1) suggests that Maine had one of the highest increases in stocking in proportion to the amount of forestland, following national trends in which forest area with a RD greater than 0.6 increased five fold (Woodall and Weiskittel, 2021). Forests with a high RD are at an increased risk for catastrophic damage from disturbances such as fire, insects and disease, drought, and windthrow (Williams et al., 2016). Disturbances reduce forest carbon stocks in the short term, inciting implications on institutional and governmental GHG emissions tracking.

The role of Maine’s forest economy and its interactions with forest dynamics have implications for Maine’s climate initiatives and policies. The Forest Opportunity Roadmap or the “FOR/Maine” initiative created by the current legislative council seeks to increase the size of the state’s forest product sector by 40% by 2025 by funding the expansion of paper mills, cross-laminated timber plants, and biomass plants (FOR/Maine, 2018). Furthermore, an executive order issued by Governor Mills specifies that Maine is to be carbon neutral by the year 2045 and reduce gross GHG emissions 80% of gross GHG emissions in 1990 by the year 2050 (An Act To Promote Clean Energy Jobs and To Establish the Maine Climate Council, 2019). The goal of ‘FOR/Maine’ in conjunction with the executive order on carbon neutrality creates tension in which short-term investments in wood production potentially conflict with long-term goals to decrease emissions and increase terrestrial carbon storage.

Assuming Maine’s forest sector maintains a static carbon sink consistent with recent levels in addition to projected declines in GHG emissions, carbon neutrality could be
reached as early as 2033 as shown in Figure 1 (Maine DEP, 2020; Forest Carbon Task Force, 2021). Considering that Maine’s forest sequestration rate have declined 5% from their 20-year peak in 2015, this trend may impact Maine’s maintenance of carbon neutrality over the long term (Domke et al., 2021). In addition, the current state of the low-grade wood product market dissuades landowners from furthering their investments in improved forest management practices (Kline et al., 2009). To correct for this market failure, stakeholders may turn to carbon markets to achieve net-zero carbon emissions for the state of Maine.

Carbon markets incentivize landowners to implement forest management practices with the goal of increasing aboveground forest carbon stocking higher than a regional aboveground CO₂ stocking baseline, ensuring that a positive net change in carbon sequestration (World Bank Group, 2020). This is also referred to as additionality’.

Figure 1. Historical and projected net GHG emissions and mitigation from the forest ecosystem and harvested wood products

Figure 1. Historical and projected net GHG emissions and mitigation from the forest ecosystem and harvested wood products
Landowners enroll their forestland in a forest carbon offset project in which additional sequestration is monitored periodically and sold in the form of a credit equivalent to 1 metric ton of CO$_2$e (World Bank Group, 2020). Compliance markets require a reduction in emissions for firms under law by means of a cap-and-trade system (World Bank Group, 2020). In comparison, voluntary markets exchange offsets for individuals and firms whose emissions are not regulated but seek to neutralize their emissions to meet emission reduction goals. Current credit prices for voluntary markets are valued between $10$-$12/tCO$_2$e (Ecosystem Marketplace, 2020). The 2021 average auction clearing price for U.S. compliance markets was valued at $9/tCO$_2$ for RGGI (Regional Greenhouse Gas Initiative) and $22/tCO$_2$e for the ARB (California-Quebec Air Resources Board) (ARB, 2022; RGGI, 2022). Compliance credits can have a higher value than voluntary markets if institutions enforce stringent regulatory reviews and verification processes, in addition to a mandatory project length of 100-years (as opposed to 40-50 years with some voluntary projects) (World Bank Group, 2020). Managing forests to maximize CO$_2$ storage and sequestration as opposed to timber revenue is a trade-off made possible by voluntary and compliance carbon markets.

Many landowners, especially small landowners, are hesitant to enroll in forest carbon offset projects due to high transaction costs and delayed returns on investment (Galik et al., 2012). These transaction costs include the costs of project design, certification, monitoring, verification, and enforcement (Dudek and Wiener, 1996). A study examining transaction costs across forest project design and accounting methodologies found that for projects with 100 hectares or less make up about 25% to 50% of the mean implementation cost per credit as opposed to 10% or less for projects with 1,000 hectares or more (Galik et al., 2012). As of 2020, 353,911 acres are enrolled in carbon markets, or less than 3% of all of Maine’s forestland (Truesdale, 2020). Maine’s carbon markets, if implemented across all 17 million acres, hold an estimated value worth over $90 million annually under the average voluntary market price in 2019 (Truesdale, 2020). As the price of carbon becomes
increasingly competitive and as more states enact climate mitigation goals, policymakers are inquiring about subsidizing forest management practices to help small landowners overcome barriers to entry in the carbon market. Small and industrial landowners alike are looking to enroll in carbon markets as the price of carbon becomes a strong substitute for low-grade wood markets (Truesdale, 2020). Stakeholders have expressed concern that enrolling more private land in carbon markets will diminish the supply of harvestable timber, negatively impacting the local economy and displacing harvests (and harvest revenue) to other regions outside of Maine in a phenomenon known as ‘leakage’ (Pan et al., 2020). Other landowners inquire about the cost effectiveness of certain management practices over the long lifetime of the project as management costs and revenues accrue at different times across the length of the project (Galik et al., 2012). Policymakers, landowners, and managers alike are looking for clear guidelines that identify cost-effective management strategies that maintain rates of historical harvesting to the mid-century and beyond (Forest Carbon Task Force, 2021).

The purpose of this thesis is to build a statewide framework to determine the cost-effective suite of forest management practices and corresponding carbon (CO$_2$) sequestration potential relative to socioeconomic and policy constraints. We take a three-pronged approach, each with its own set of inputs, parameters, skill level requirements, and outcomes. In the first approach, we review a group of biophysical, spatial, and silvicultural studies in the northeast to determine changes in net forest carbon sequestration across different treatments and harvesting practices. This allows us to estimate additional sequestration above a baseline, the cost of mitigation, and determine the strength of Maine’s forest carbon sink over various levels landowner participation. Secondly, we analyze forest growth and yield projections (and subsequently carbon sequestration) across 6 different management regimes from the Forest Vegetation Simulator (FVS) and determine the net present value (NPV) of each pathway across 100 years. Thirdly, we construct an optimization model in Woodstock using Model I and Model II
approaches that maximize NPV under various policy and economic constraints across forest management regimes and treatments. Combined, the three approaches create a modeling framework that will facilitate in developing a more precise forecast over a 100-year horizon, allowing us to determine the suite of management practices that minimize the societal cost of enhancing Maine’s forest carbon sink (Daigneault et al., 2021).

This paper is organized as follows: first, we present the methods of our literature assessment of net carbon sequestration resulting from changes in forest management practices in addition to their mitigation cost. Next, we present the methods of our NPV study using outputs from the FVS model over 6 different management regimes, followed by the methods of our Woodstock linear programming model, using both Model I and Model II approaches. In section 3, we present the results of our literature assessment, FVS study, and Woodstock model respectively. We conclude this paper with a synthesis of our findings and suggestions for future research.

2 METHODS

To estimate the net forest carbon sequestration potential and net present value across different management practices and regimes, we use three different strategies; a literature assessment, a forest growth and yield simulation using the Forest Vegetation Simulator (FVS), and two forms of linear program optimization models using Woodstock. The first approach is not a modeling approach; rather, we conduct a in-depth literature review to estimate the net forest carbon sequestration potential and cost of mitigation in Maine across various practices and implementation rates. The second and third strategies are both forms of forest modeling and achieve the same goals, albeit with different levels of accuracy and detail. All three approaches present conclusions about Maine’s forest mitigation potential over a 100-year time horizon while accounting for the net present value or break-even cost of each management practice or suite of practices across Maine’s 17,000,000 acres of working forests.
2.1 Literature Assessment and Analysis

To estimate the net forest carbon sequestration potential across different management practices and regimes, we first examined 9 studies and collected over 100 observations of changes in carbon flux resulting from silvicultural implementations or modeling simulations that take place in the northeastern U.S. Management practices from these study range from a low-intensive ‘no management’ scenario where no biomass is harvested, to a high-intensive clearcut harvest scenario where nearly all of the basal area is removed. Negative carbon flux rates indicate sequestration from the atmosphere into the forest, whereas positive fluxes indicate CO$_2$ emissions. We convert fluxes to a standard rate which represent a ton of carbon dioxide on a per acre, per year basis (tCO$_2$/ac/yr). Fluxes were then compared to the study-specific baseline. Nearly all studies assume a constant climate reflecting historical trends in forest growth and yield, but vary widely in projection timelines, model modalities, sample size, and assumptions regarding harvest trends and the amount of carbon stored in harvested wood products (HWP). To compare the sequestration potential across management practice categories, we averaged the additional flux rates. The publications, along with their simulated forest management practices and findings, are summarized in Table 1.
<table>
<thead>
<tr>
<th>Study</th>
<th>Study Location</th>
<th>Study Area (acres)</th>
<th>Model Years</th>
<th>Baseline</th>
<th>Method/Model</th>
<th>Management Assumptions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Daigneault et al. (2021)</td>
<td>ME</td>
<td>9,000,000</td>
<td>(2020-2100) 100 years</td>
<td>90% harvest from partial cut; 10% from clearcut 30-year sequestration average</td>
<td>LANDIS-II Forest Landscape Model</td>
<td>avoided conversion, CC +plant, extended rotation, no management</td>
</tr>
<tr>
<td>Dugan et al. (2021)</td>
<td>VT</td>
<td>4,500,000</td>
<td>(2020-2050) 30 years</td>
<td>Land use change conversion</td>
<td>CBM-CFS3 and CBM-FHWP</td>
<td>afforestation, extended rotation, PCT, uneven-age management, no management</td>
</tr>
<tr>
<td>Cook-Patton et al. (2020)</td>
<td>ME (U.S.)</td>
<td>370,143</td>
<td>(2020-2050) 30 years</td>
<td>Land use change conversion</td>
<td>Spatial analysis using ArcMap v10.3.1, Python, &amp; R</td>
<td>afforestation, increase stocking</td>
</tr>
<tr>
<td>Gunn &amp; Buchholz (2018)</td>
<td>ME</td>
<td>7,500,000</td>
<td>(2010-2110) 100 years</td>
<td>Business as usual</td>
<td>ForGATE</td>
<td>CC+ natural regen, PCT, uneven-age management, no management</td>
</tr>
<tr>
<td>Ford &amp; Keeton (2017)</td>
<td>VT, NY</td>
<td>60</td>
<td>(2006-2016) 10 years</td>
<td>No management scenario</td>
<td>Experimental treatments with FVS modeling for no management</td>
<td>even-age management, extended rotations</td>
</tr>
<tr>
<td>Nielsen et al. (2014)</td>
<td>ME (U.S.)</td>
<td>957,450</td>
<td>(2005-2090) 85 years</td>
<td>Land use change conversion</td>
<td>Cost transfer analysis using NCLD and CRP data</td>
<td>afforestation</td>
</tr>
<tr>
<td>Russell-Roy et al. (2014)</td>
<td>VT</td>
<td>966</td>
<td>(2010-2110) 100 years</td>
<td>High grading</td>
<td>FVS with Climate Action Reserve Forest Project Protocol 2.1</td>
<td>CC+ natural regen, even-age management, uneven-age management, no management, PCT</td>
</tr>
<tr>
<td>Hoover &amp; Heath (2011)</td>
<td>ME, VT, NH, MA, CT, NY, PA</td>
<td>7,500,000 (ME only)</td>
<td>(2005-2085) 30, 60, 90 years</td>
<td>Maintain current levels of stocking (9.8 TG C)</td>
<td>USDA FVS and FIA Mapmaker 2.1</td>
<td>increase stocking</td>
</tr>
<tr>
<td>Nunery &amp; Keeton (2009)</td>
<td>ME, VT, NH, NY</td>
<td>160,000</td>
<td>(2000-2120) 15, 30, 80, 120 &amp; 160 years</td>
<td>No management scenario</td>
<td>USDA FVS</td>
<td>CC+ natural regen, even-age management, uneven-age management</td>
</tr>
</tbody>
</table>

Table 1. Comparison of carbon sequestration studies in the northeastern U.S.
For part of our literature assessment we collected the ‘break-even’ carbon price and changes in regional harvesting levels of each practice if reported within the study. The ‘break-even’ carbon price can be thought of the price the landowner is willing to accept to change their behavior (usually from a lesser to a greater sustainable practice) to implement a certain practice measured in $/tCO₂e (Daigneault et al., 2021). Assessing the relationship between projected changes in harvest levels and carbon price relative to changing sequestration rates shapes our understanding regarding the economic trade-offs landowners face when making forest management decisions. From the nine studies analyzed, only three studies estimated changes in baseline harvest levels from imposing different management regimes in their study area (Gunn and Buchholz, 2018; Daigneault et al., 2021; Dugan et al., 2021) and four included the cost of mitigation (Cook-Patton et al., 2020; Daigneault et al., 2021; Russell-Roy et al., 2014).

Based on LANDIS-II model projections, clearcut harvests followed by artificial regeneration decrease baseline harvesting levels an average of 7% per year over 20 years and increase levels 1% on average over 50 years, with a break-even cost of $15/tCO₂e and $10/tCO₂e respectively (Daigneault et al., 2021). The variation in harvesting levels is attributable to the current stand conditions in the study area in which many stands are eligible for harvest in the short term, meaning few are eligible over the long term. This trend is also observed when extending the minimum age of harvest from 50 to 85, as harvest levels decline 44% in the first 20 years and only 10% when standardized over 50 years (Daigneault et al., 2021). The high opportunity cost of extending rotations reflect losses in harvest revenue, with a projected break-even cost between $17-$22/tCO₂e (Daigneault et al., 2021). In contrast from the previous study, clearcut harvests followed by natural regeneration project annual harvest yields to decline 11% to 15% per year when averaged over 300 years (Gunn and Buchholz, 2018). A similar drop in harvests occur when extending rotations from a minimum age of 80 to 90 (-10%) (Dugan et al., 2021).
The only practices resulting in increased harvesting levels were uneven-age management targeting a 30% removal of the basal area every 30 years (Gunn and Buchholz, 2018).

The break-even cost of converting non-forest land into forest land (afforestation) in Maine is projected to be $40/tCO$_2$e on average when examining conversion from urban-open areas where land rent is high, to $15 when examining the cost of conversion from pasture and shrub land only (where land rents are much lower) (Cook-Patton et al., 2020); Nielsen 2014). The previous studies have used landscape-level modeling to determine the cost of mitigation, but Russel-Roy et.al. uses a stand-level model to determine a break-even cost that reflects the difference between revenues earned from harvests and revenues earned from carbon credit markets. When finding the break-even cost of various management regimes over 100 years, the average break-even costs occur of clearcut harvesting is $20/tCO$_2$e/year and $14/tCO$_2$e/year for shelterwood harvests (Russell-Roy et al., 2014). Next in our analysis, we use these figures to determine the average break-even cost for each of the nine forest carbon practices.

**Statewide Carbon Sequestration Potential**

After grouping the observations, we devised a statewide assessment of the potential net carbon sequestration benefits and economic costs across 9 studies, and used it to inform stakeholders about the sequestration potential of Maine’s forests. The Forest Carbon Task Force, a subcommittee informing the Maine Climate Council, was tasked with recommending a numeric goal or target for increased carbon sequestration in Maine by 2050 (Maine Climate Council, 2020). Central to this assessment was identifying the suite of forest management practices that sequester the most CO$_2$ from the atmosphere and their respective costs. Our literature analysis aided the task force by synthesizing the estimated ‘additional’ sequestration rate (tCO$_2$/ac/yr) and break-even cost across the state’s entire forest landscape. Table 2 details the treatment allocation across 9 management practices. We detail the methods of our assessment below.
<table>
<thead>
<tr>
<th>#</th>
<th>Scenario</th>
<th>Acres (ac)</th>
<th>CC &amp; Natural Regen</th>
<th>CC &amp; Artificial Regen</th>
<th>Increase Stocking</th>
<th>Extend Rotations</th>
<th>Uneven-age Harvest</th>
<th>Even-age Harvest</th>
<th>PCT</th>
<th>No Management</th>
<th>Avoided Conversion</th>
<th>TOTAL</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td><strong>Baseline</strong></td>
<td>3,986</td>
<td>16,479</td>
<td>5,408</td>
<td>-</td>
<td>-</td>
<td>182,917</td>
<td>182,917</td>
<td>9,495</td>
<td>112,000</td>
<td>10,000</td>
<td>530,100</td>
</tr>
<tr>
<td></td>
<td>%</td>
<td>1%</td>
<td>3%</td>
<td>1%</td>
<td>-</td>
<td>-</td>
<td>35%</td>
<td>35%</td>
<td>2%</td>
<td>21%</td>
<td>2%</td>
<td>100%</td>
</tr>
<tr>
<td>#1</td>
<td><strong>CC + plant 50% of area annually</strong></td>
<td>3,986</td>
<td>-</td>
<td>265,050</td>
<td>-</td>
<td>-</td>
<td>64,785</td>
<td>64,785</td>
<td>9,495</td>
<td>112,000</td>
<td>10,000</td>
<td>530,100</td>
</tr>
<tr>
<td></td>
<td>%</td>
<td>1%</td>
<td>-</td>
<td>50%</td>
<td>-</td>
<td>-</td>
<td>12%</td>
<td>12%</td>
<td>2%</td>
<td>21%</td>
<td>2%</td>
<td>100%</td>
</tr>
<tr>
<td>#2</td>
<td><strong>Increase stocking levels; convert all CC to CC + plant</strong></td>
<td>3,986</td>
<td>-</td>
<td>21,877</td>
<td>150,030</td>
<td>-</td>
<td>97,361</td>
<td>97,361</td>
<td>28,485</td>
<td>112,000</td>
<td>10,000</td>
<td>10,000</td>
</tr>
<tr>
<td></td>
<td>%</td>
<td>1%</td>
<td>-</td>
<td>4%</td>
<td>30%</td>
<td>-</td>
<td>19%</td>
<td>19%</td>
<td>5%</td>
<td>21%</td>
<td>2%</td>
<td>100%</td>
</tr>
<tr>
<td>#3</td>
<td><strong>Afforestation (x4); convert all CC to CC + plant</strong></td>
<td>12,338</td>
<td>-</td>
<td>21,887</td>
<td>-</td>
<td>-</td>
<td>182,190</td>
<td>182,190</td>
<td>9,495</td>
<td>112,000</td>
<td>10,000</td>
<td>530,100</td>
</tr>
<tr>
<td></td>
<td>%</td>
<td>2%</td>
<td>-</td>
<td>4%</td>
<td>-</td>
<td>-</td>
<td>35%</td>
<td>35%</td>
<td>2%</td>
<td>21%</td>
<td>2%</td>
<td>100%</td>
</tr>
<tr>
<td>#4</td>
<td><strong>Extend rotations on 50% of the landscape</strong></td>
<td>3,986</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>265,050</td>
<td>55,290</td>
<td>55,290</td>
<td>28,485</td>
<td>112,000</td>
<td>10,000</td>
<td>530,00</td>
</tr>
<tr>
<td></td>
<td>%</td>
<td>1%</td>
<td>-</td>
<td>-</td>
<td>50%</td>
<td>-</td>
<td>11%</td>
<td>11%</td>
<td>5%</td>
<td>21%</td>
<td>2%</td>
<td>100%</td>
</tr>
</tbody>
</table>

Table 2. Application of management practices across the baseline and 4 scenarios
We used the Maine Silvicultural Activities Report to determine the maximum number of acres eligible for each management practice based on historical trends concerning the amount of acres harvested under different practices annually, totaling 530,000 acres (MFS, 2015). In addition to the nine practice groups, we add 'avoided conversion' in which the current forest area is held constant at the cost of renting the land at the cost of the best and highest use converted, at a rate of 212 tCO$_2$/ac and cost of $17/year (Daigneault et al., 2021). This is not a silvicultural practice on its own, however its necessary to implement in a landscape scale assessment as it represents potential of avoided’ actions to increase net sequestration (Daigneault et al., 2021). This analysis gives us a baseline scenario to determine the point of net-zero mitigation. We then apply a range of participation rates which represent the amount of acres treated to find the bounds of mitigation potential across all of Maine’s 17,000,000 acres in one year.

After creating the baseline, we form 4 additional scenarios; clearcut and plant, increase forest stocking, quadruple the amount of afforestation, and extend rotations. To do this approach, we expand the amount of acres treated annually for a set of practices above the baseline for carbon-sequestering practices. Table 2 details the treatment allocation across 9 management practices. Total acres are constrained by the maximum amount of acres eligible for conversion from non-forest land to forestland (370,143 acres) (Cook-Patton et al., 2020) in addition to the constant, non-cumulative rate of 10,000 acres per year for avoided conversion, equivalent with pressures of land rent (Daigneault et al., 2021). We examine statewide trends in sequestration using this data, assuming linear growth, across 100 years. Lastly we use average break-even costs of mitigation on a per acre per year basis to estimate the total cost of mitigation.

2.2 FVS Modeling and Analysis

The previous literature assessment allows us to estimate the sequestration potential of Maine’s forest under different management practices; however, rates of sequestration are
nonlinear just as forest growth rates are nonlinear. Standardizing CO$_2$ flux rates over the length of the study period dampens our ability to determine the inter annual variability spanning a century. Additionally, many forest managers note that long-term forest management occurs not through iterations of the same practice, but through a suite of management practices implemented in chronological order known as regimes (Russell-Roy et al., 2014). Table 1 presents four studies that examine how applied forest management regimes change growth, yield, and aboveground CO$_2$ stocking overtime using a biophysical forest modeling system known as the Forest Vegetation Simulator (FVS) through the USDA (Russell-Roy et al., 2014; Keeton and Ford, 2017; Nunery and Keeton, 2010; Hoover and Heath, 2011).

FVS is a distance-independent, individual-tree growth model that forms growth projections derived from current forest composition conditions, and yield projections derived from harvest protocols (Crookston and Dixon, 2005). After modeling individual-tree growth, FVS aggregates the tree-level output to a ‘stand-level’ output using expansion factors (Crookston and Dixon, 2005). If the surveyed stand meets a conditional threshold, for example a factor of relative density or basal area, a treatment is applied systematically over a user-defined time horizon (Crookston and Dixon, 2005). FVS is widely used by both forest managers, planners, and carbon market developers for estimating carbon sequestration in managed forests as part of climate change mitigation projects (Keeton and Ford, 2017). Additionally, the program is accessible to the general public online compatible with USDA’s Forest Inventory Analysis (FIA) data, and contains regional variants that can be calibrated to represent stand composition across a dozen regions in the U.S. (Crookston and Dixon, 2005)

In order to investigate changes across pathways for carbon sequestration in the state of Maine, we use 100 year projections (2019-2119) in five year periods from FVS to determine the optimal silvicultural regime over that maximizes the present value of carbon storage and timber harvests at year 100. For our analysis, we input 3,490 Maine FIA plots
collected over a five-year horizon from 2015-2019, ensuring that we analyse the most
up-to-date forest conditions. Each plot has a respective "weight" which represents a share
of Maine's 17 million acre working forests and each regime is applied statewide scale such
that the entire state adopts a particular regime (Crookston and Dixon, 2005). We assume
constant climatic conditions and make no assumptions regarding changes in disturbance in
the simulations.

In FVS we use the component or 'base model' with the Acadian geographic variant to
predict the growth and yield of individual trees under 6 applied management regimes
presented in Table 3. (Crookston and Dixon, 2005). The Acadian variant serves as a
regional lens to FVS, depicting greater specificity in growth and yield trends of spruce and
fir trees by means of applying series specific equations for crown width, total height, height
to crown base, diameter and height increment, and mortality (Weiskittel and Kershaw,
2018). The regimes in Table 3 reflect the diverse management approaches used by forest
managers throughout the state. Each silvicultural regime (not including 'BASE') is
comprised of 2 or more treatments occurring when the stand meets an initial condition, 10,
and 20 years after the triggered action. In this approach, there are no external constraints
to limit actions triggered by the simulation model, only biophysical constraints. Because
we aggregate stands up to a state-wide landscape, all stands meeting a basal area or
stocking condition can trigger harvest in one period that is up to 10 times the historic
harvest maximum for the state. In other words, FVS produces output based on the
biophysical parameters only, and it does not implement any economic or policy constraints
that would otherwise reduce harvest volumes in any period.

After obtaining the FVS output, we apply a set of calculations across carbon pools and
revenue streams before we can determine the total NPV for each of the 6 management
regimes. FVS output retains distinctions between each plot's forest type $f$ and region $r$ in
addition to the amount of saw wood and pulp wood $i$ for standing and harvested wood.
These components are endogenous with the exception of exogenous carbon prices $z$. Rather
Table 3. Silvicultural regimes and treatments simulated in FVS

than use the break-even carbon price to determine the cost of sequestration as in the literature assessment, we apply carbon prices endogenously across low, medium, and high values. For our analysis, the 2019 CO$_2$e live stocking in the business-as-usual scenario (BAU) represents the 'baseline' stocking level. Carbon revenues are rewarded with each periodic gain in stocking above the baseline. In this way we mirror the logic of carbon credit registries (Daigneault and Strong, 2019). For consistency, we apply a penalty equivalent to the price of carbon for every ton that drops below the baseline or the prior period. The 'low' and 'medium' exogenous carbon prices represent the average auction settlement price between Q1 2019 and Q4 2021 valued at $7/CO_2$e for RGGI and $20/CO_2$e for ARB (ARB, 2022; RGGI, 2022). We also apply a 'high' carbon price scenario of $51/CO_2$e, which represents the social cost of carbon (SCC) and often serves as an upper-bound in forest mitigation modeling (Bluffstone et al., 2017). We denote these attributes or 'components' in Table 3.

These components vary by parameters in Table 4, illustrating, for example that the average stumpage price $P_H$ differs by forest type and region. Our growth and yield outputs denote the amount of harvested timber $H$ and the amount of standing timber $Q$ in each 5-year time period $t$. Carbon storage factors were applied as a factor by $f$ for standing wood and $i$ for harvested wood. The factor represents the quotient between aboveground carbon in live trees and the net merchantable volume of growing stock trees in cubic feet, using FIA data. We determine the amount of tCO$_2$ in standing live merchantable trees and
in harvested wood products using a mean decay rate of 20% (Li et al., 2022). By factoring in CO₂ stored in HWP, we reward landowners for producing large saw timber which retains more carbon in long-lived, durable wood products.

For each regime, we determine the total NPV comprised of 3 sources: harvested merchantable timber ($NR_H$), carbon stored in HWP ($HWP$), and live aboveground CO₂e ($AGC$). To find the net revenue of timber harvests, we use both the stumpage price $P_H$
represented as dollars per thousand board feet ($/MBF) and $/gt of pulpwood using the 5 year average price (denoted by region and species) from the Maine Stumpage Price Report (MFS, 2015). We subtract the cost of implementing planting and pre-commercial thinning practices $E$ to reflect costs not reflected in the take-home stumpage price. We use determine that the regeneration cost of 800 trees-per-acre (TPA) of saplings averages $550 per acre, as used in the LANDIS-II model of northern Maine (Daigneault et al., 2021) and a cost of $300 per acre for pre-commercial thinning as presented from Weyehauser, 2021 and Daigneault et al., 2021.

\[ AGC = Q_{fr} \alpha_f \]  

\[ HWPC = \beta_i H_{if} \]  

\[ NR_H = P_{ifr} H_{izr} - E_m \]  

Next we calculate the revenue streams from carbon sequestration and HWP storage. Carbon flux is derived from a change in standing aboveground carbon \(AGC\) from period \(t\) to period \((t-1)\) can be thought of as the carbon credits awarded for increased mitigation across both standing timber and harvested wood products, valued in $/tCO_2e. The approach can also be thought of the equivalent of receiving a payment to the yearly change in carbon sequestration, and is a method commonly used when provided carbon stocking projections only (Daigneault and Strong, 2019). Similarly, every tCO\(_2\)e lost between each period receives a penalty equal to the carbon price \(z\), following the logic of carbon crediting programs.

\[ NR_C = P_c \times (AGC_t - AGC_{t-1} + HWPC_t) \]
By taking the sum of carbon and harvested timber revenue streams, we calculate NPV by applying the discount rate for time $t$. Natural climate solutions valuation often applies a yearly discount rate between 2.5% and 7% (Carle et al., 2021); therefore, we apply rates an average discount rate of 5% to serve as a benchmark in which we will compare the total NPV from 2019 to 2119. Further, we conduct a sensitivity analysis using 3%, and 7% discount rate, allowing us to evaluate the interactions between harvest timing and intensity across the 6 regimes.

$$NPV = \sum_{t=1}^{T} \frac{NR_H + NR_C}{(1 + \delta)^t}$$ (5)

In the results section 3.2, we compare the total NPV across the 6 management regimes and carbon price scenarios for 100 years after presenting the results of our FVS simulated output. Lastly, we conduct a sensitivity analysis to compare how changes in the discount rate affect management regime rankings.

### 2.3 Strategic Modeling Using Woodstock

In section 2.1, the nine selected studies use one or more of the following three techniques: forest stand simulators, spatial empirical relationships on a landscape level, or silvicultural application. Silvicultural applications are real-world experiments whose findings support simulations and empirical relationships on forest growth and yield (Keeton and Ford, 2017). Stand-level forest simulation models such as FVS have the ability to project changes in forest growth and yield based on biophysical forest conditions (Crookston and Dixon, 2005). In section 2.2, we outline our approach to disseminating changes in net sequestration and net present value on a landscape level from FVS output. However, one key limitation of this approach it that it does not apply economic, policy, and operational constraints nor set economic objectives that would otherwise alter the
management decisions implemented across the landscape. Such flexibility and computational complexity is necessary to examine dynamic forest management decisions across a high-level landscape and long time horizon (Gunn, 2007).

Complex forest sector modeling can be accomplished through a variety of techniques to examine changes in forest growth and yield, land use, mitigation and emissions across economic and policy scenarios on a landscape scale. Previous research by Daigneault et al. (2021) use the ecological landscape model LANDIS-II to capture the break-even carbon price across seven simulated forest management practices in northern Maine forests. They estimate that carbon sequestration can increase by 2 MMtCO$_2$e/year through a mix of set asides, clearcut and planting, and partial harvesting treatments (Daigneault et al., 2021). Similarly, an econometric modeling approach was taken to examine changes in aboveground carbon stocks over 100 years in the northeast using area change and harvest probability as the dependent variable using the Shared Socioeconomic Pathways (SSPs) scenario framework (Zhao et al., 2022). Results show that the total forest sector carbon (AGC and HWP) rise by 0.40- 0.64% annually by from 2020 to 2100, with the HWP carbon pool increasing as prices for large-diameter timber increase (Zhao et al., 2022). However within development-centered SSP1 and SSP5, rates of carbon sequestration decline throughout the century by 1% in Maine (Zhao et al., 2022).

The previous modeling approaches do not determine net present value (NPV) as dependent variable across shifts in silvicultural practices and carbon policy scenarios. With exception, a New Hampshire study analyzes the NPV of standing and harvested timber regime over high and low carbon pricing pathways over 100 years (Gutrich and Howarth, 2007. Their findings indicate that the NPV of this scenario for traditional rotation age or length of harvest is $2438 per acre more than the NPV from timber alone by any means of forest management or avoided conversion. (Gutrich and Howarth, 2007). However this analysis occurs only at a stand-level and is therefore limited by fixed assumptions in
product uniformity, forest composition, and forest characteristics (Gutrich and Howarth, 2007).

Other forms of advanced forest analysis via the Global Timber Model (GTM) or the Forestry and Agricultural Sector Optimization Model (FASOM) employ structural dynamic optimization modeling to determine the economic returns across forest-management and ecosystem management decisions in addition modeling changes in forest carbon stocks from trends in growth, yield, and forest disturbances (Austin et al., 2020; Wear and Coulston, 2019). The strength of these advanced models lie in their ability to optimize forest carbon sequestration on a global and regional scale over long periods of time, and their integration of complex, non-linear functional forms to solve iterative market and socioeconomic relationships (M. et al., 2019). These models often use large-scale spatial inputs, spanning several states or nations, and can therefore create stronger assertions relating to national mitigation and emissions policy (M. et al., 2019). Dynamic, non-linear programming (NLP) models incorporate large-scale practices such as replanting, creating no harvest reserves, and harvesting endogenously to maximize the NPV or determine the policies that minimize the social cost of achieving sequestration goals (Austin et al., 2020).

Like the NLP dynamic optimization models, dynamic linear programming (LP) models have the ability maximize (or minimize) one objective, along with linearly-related constraints to reflect landowner or state ‘decisions’ (Gunn, 2007). Often used by land managers, planners, and governments, these models merge the forest dynamics of biophysical models with economic demand models for timber and timber substitutes, extending researchers ability to make long term assumptions about policy and market activity (Wear and Parks, 1994; Sohngen, 1998). In this way, the model processes forest growth and harvest scheduling relative to certain economic, policy, and operational constraints, such as the minimum number of acres left untreated in a given year. The strength in this modeling approach lies in its ability to analyze shadow costs across complex choice sets, allowing the model to analyze all options and select only a specific
combination of which maximizes a certain goal (Gunn, 2007). However, the accuracy of LP models hinge on its ability to incorporate disaggregated forest conditions using plot level inputs, and determine changes in growth and yield based on biophysical changes in a stand-level (Gunn, 2007). In other words, large-scale, NLP models homogenize forests, as opposed to regionalized or local LP models which evaluate aggregate changes using optimization functions on a plot-level.

When the focus is on modeling regionalized changes in management across the forest landscape, two types of LP models are commonly used: Model I and Model II which are illustrated in Figure 2 (Gunn, 2009). Model I is less computationally complex than Model II in that it contains decision variables that choose from a set of regimes to be applied to specific stands, with each regime running the entire length of the model (Martin et al., 2017). Looking forward in time, the model selects the optimal regime curve that maximizes the given objective function (Gunn, 2007). Model I requires growth and yield curves for each individual plot, and is commonly thought of as a ’spatial’ model, as growth rates change following each iteration of silvicultural treatment along each growth and yield curve (Gunn, 2009).

In contrast, Model II is thought to be ’aspatial’ in which stands of similar composition, class, and characteristics are aggregated along one or more growth curves, allowing the model to determine the optimal timing of silvicultural treatments (Martin et al., 2017). In this way, Model II has the ability to consider many prescription alternatives without having to define them explicitly, allowing an additional layer of flexibility when maximizing objective functions. In Model II yield curves, growth is largely a function of the average age of the stand, displaying the early, middle, and late phases of forest growth as opposed to the predetermined biophysical simulations of growth and yield under Model I. In this way, Model II is more computationally complex than Model I as it considers an additional choice to the matrix concerning the optimal harvest time of any treatment defined in the model (Martin et al., 2017).
Woodstock is a commercialized LP solver, often used by forest managers and planners, that allows users to initialize the forest inventory, mimic silvicultural prescriptions, set management objectives and constraints, and build alternative scenarios to explore trade-offs of different scenarios (Gunn, 2009). This program is able to model using both Model I and Model II approaches. One North Carolina study uses a Model II approach in Woodstock to determine the production possibility frontier for the NPV of carbon sequestration and timber resources over 100 years in a 59,280 acre research forest (Roise et al., 2016). When constraining the flow in harvests by 20%, the model selected stands with the highest planting density and extended harvest rotations, yielding an additional 5.6 MMtCO$_2$e of carbon stored at a NPV of $48.5$ million, with an opportunity cost of $2.34$/tCO$_2$e (Roise et al., 2016). When maximizing timber resources under the same harvest constraints, the NPV increases by $13$ million, with no additional sequestration, leading the authors conclude that opportunity costs associated with increasing carbon sequestration are less than the 2015 California carbon market price of $10$/tCO$_2$e for their study site (Roise et al., 2016).
Another analysis used the Model II approach in Woodstock to examine the influence of pre-commercial thinning, herbicide application, and clearcut with artificial regeneration on Maine’s future wood supply when maximizing NPV at a 6% discount rate from 1995 to 2095 (Wagner et al., 2010). Yield curves and stumpage prices for this model were developed empirically and aggregated across 3 forest types: hardwood, softwood, and planted spruce/pine stands (Wagner et al., 2010). Furthermore, constraints imposed in the model included historical trends in treatment area, steady harvest volume (2.8 billion ft$^3$ every 5 years), and an even harvest flow constraint (Wagner et al., 2010). The model allocated 15% of treatment acres to herbicide application, 85% to a pre-commercial thinning regime, and no acres for the clearcut and planting treatment in order to maximize the total NPV valued at $617 million (Wagner et al., 2010).

This thesis employs the use of a dynamic optimization LP model using the Remsoft Woodstock program to maximize a range of objectives subject to 2015-2019 statewide trends in harvest intensity and silvicultural activity via the Maine Silvicultural Activity Report (MFS, 2015). Due to the regionalized scope of our analysis, the availability of small-scale stand growth and yield projections via FIA data, and our ‘position’ as landscape-scale forest planners, the LP modeling approach is effective choice. To capture spatial, biophysical precision that the Model I approach provides while also incorporating an additional layer of dynamic harvest choice under the Model II approach, we use both methods to determine the suite of regimes that maximize 8 different objective functions relative to harvest and area constraints across the state. Henceforth, we refer to Model I as a ‘regime based approach’ and Model II as a ‘treatment based approach’. Figure 3 provides a schematic of our dynamic optimization LP model using both Model I and Model II. The operability of Model I and Model II vary; therefore, we discuss the integration of management practices, objectives, and constraints for each.
Model I: Regimes Based Approach

We input the BASE, BAU, CCPL, CTH, COVR, and PCCT regime growth and yield outputs and their respective area weights into Woodstock for 2,958 forested FIA plots. Further, we integrate the same equations from section 2.2 for AGC, HWPC, NR_H, and NR_C. A notable difference is that the parameters of forest area X, harvest quantity H, and standing inventory Q (displayed in Table 5) become variables rather than parameters, as they change according to the objective and actions of the model. For Model I, we let X_{mt} represent the number of acres X undergoing a certain management regime m in period t,
Table 6. Model I and Model II variables

<table>
<thead>
<tr>
<th>Variable</th>
<th>Notation</th>
</tr>
</thead>
<tbody>
<tr>
<td>forest area</td>
<td>$X$</td>
</tr>
<tr>
<td>harvest quantity</td>
<td>$H$</td>
</tr>
<tr>
<td>forest inventory</td>
<td>$Q$</td>
</tr>
</tbody>
</table>

where regime $m$ has a specific rotation length and set of treatments that are repeated for the duration of the planning horizon.

To compare across policy and mitigation scenarios, we construct 8 different maximization objectives: (#1) maximize total revenue relative to a static harvest constraint, (#2) maximize total CO$_2$ stocks (from HWP and AGC pools), (#3) maximize total harvest volume (in merchantable ft$^3$), (#4) maximize total net revenue, (#5) maximize total NPV from harvest revenues, (#6) maximize the NPV under the RGGI carbon price scenario, (#7) maximize the NPV under the ARB carbon price scenario, (#8) maximize the NPV under the SCC carbon price scenario. For all of the NPV scenarios we first use a discount rate of 5%. All objectives are run with only the total area constraint (#), harvest volume constraints only (#.a) and harvest volume plus treatment area constraints (#.b) with the exception of our baseline scenario (#1b) which follows a strict, 5-year average harvest level in which constraint a is static, represented as:

$$H_t(m) = 3.0 \text{ billion ft}^3$$

Our baseline scenario harvests at the 5-year average harvesting level and serves as a benchmark of which we can compare against scenarios 2 - 8. Table 6 displays the structure of our scenarios.

The total forest area constraint where $X_{mt}$ must equal 17.1 million acres in each period, is automatically integrated into the model. We assume that total forest area remains static over 100 years (no change in land use). Like the total area constraint, Woodstock also applies non-negativity constraints regarding harvest volume and inventory in which
<table>
<thead>
<tr>
<th>Constraint</th>
<th>Objective X</th>
<th>Objective X.a</th>
<th>Objective X.b</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total area (acres)</td>
<td>$X_{frm} = 17.1$ million</td>
<td>$X_{frm} = 17.1$ million</td>
<td>$X_{frm} = 17.1$ million</td>
</tr>
<tr>
<td>Minimum harvest</td>
<td>$2.90$ billion $ft^3 \leq H_t \geq 3.3$ billion $ft^3$</td>
<td>$2.90$ billion $ft^3 \leq H_t \geq 3.3$ billion $ft^3$</td>
<td></td>
</tr>
<tr>
<td>Treatment area (Acres)</td>
<td>$0$ acres $\leq H_t(BAU) \geq 3,180,000$ acres</td>
<td>$0$ acres $\leq H_t(CTH) \geq 3,180,000$ acres</td>
<td>$0$ acres $\leq H_t(PCCT) \geq 390,000$ acres</td>
</tr>
</tbody>
</table>

Table 7. Design of maximization scenarios and their constraints

stocking levels and harvests cannot be negative. The second constraint (#.a) is a minimum 5-year harvest volume of $2.90$ billion merchantable $ft^3$ and a maximum harvest volume of $3.3$ billion merchantable $ft^3$ which represents the 10-year maximum of total harvest removals in a 5-year period. Our third set of constraints (#.b) involve setting lower and upper bounds for the total area treated under the 6 regimes in a 5-year period, subject to the optimization function, detailed in Table 7. By running the model with three sets of constraints, we can determine the upper and lower bounds of harvesting for each objective.

We do not define any actions into the model in this Model I approach. Rather, the model chooses one of the six management regimes for each plot, such that the suite of management regimes satisfy the objective total and sum to $17.1$ million acres.
Model II: Treatment Based Approach

For Model II, we let $X_{mjt}$ represent the number of acres $X$ undergoing a certain silvicultural action $m$ harvested in period $t$. Unlike Model I, Model II inputs only the BASE or 'no management' growth curve from FVS and relies on the average stand age, in addition to its growth conditions, to trigger actions. In other words, the BASE curve serves at the starting point for all stands in the model which is subject to the model-defined treatments. Within Woodstock, we reconstruct regimes by manually integrating the treatments in Table 3 as actions. We use the same thresholds or conditions regarding BA, RD, and age, allowing Woodstock to trigger the action once the condition is met. This approach gives the model another dimension of flexibility, as it has the option of performing the harvest action several periods after the stand has met a conditional threshold, or not perform the action at all, subject to maximizing the objective function subject to the constraints (Gunn, 2009).

For this approach, we determine the growth trajectory across the 7 forest types by aggregating the growth response by age and forest type from the BASE or 'no management" growth curve. We aggregate the curves by finding the median volume of merchantable ft$^3$ across all average stand ages, smoothed overtime using a logarithmic function of growth and age. Figure 4 displays forest growth curves that auctioned stands transition to in Woodstock by forest type under the BASE regime from ages 0 to 60. The growth rate of a planted white-spruce stand following a clear-cut harvest also requires its own curve. This trajectory is derived from the same aggregation and smoothing process on the CCPL regime outputs.

The Woodstock interpreter assumes that actions happen at the end of every 5-year planning period, mirroring FVS projections. We are able to replicate regimes in the model by 'locking' the ability of stands to undergo shelterwood harvests and overstory removals for 10 and 20 years respectively. Furthermore, treatments must occur in chronological order, such that only stands eligible for a shelterwood harvest removing 70% of the BA
Figure 4. Functions of forest growth by forest type used to transition BASE stands in Woodstock

<table>
<thead>
<tr>
<th>Code</th>
<th>Practice</th>
<th>Stand Requirements</th>
<th>Treatment 1</th>
<th>Treatment 2</th>
<th>Treatment 3</th>
<th>Treatment 4</th>
</tr>
</thead>
<tbody>
<tr>
<td>BASE</td>
<td>No management</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>BAU</td>
<td>Business as usual</td>
<td>BA &gt;70</td>
<td>SHEL70</td>
<td>OSR</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>CCPL</td>
<td>Clearcut &amp; plant spruce</td>
<td>Merchantable saw ft² ≥ 400 ft² or average stand age ≥ 50</td>
<td>CCPL</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>CTH</td>
<td>Commercial thinning</td>
<td>RD = 0.55</td>
<td>SHEL70</td>
<td>SHEL70</td>
<td>OSR</td>
<td>-</td>
</tr>
<tr>
<td>PCCT</td>
<td>Commercial &amp; pre-commercial thinning</td>
<td>TPA ≥ 800 mt age 10 or RD = 0.55</td>
<td>PCT</td>
<td>SHEL70</td>
<td>SHEL70</td>
<td>OSR</td>
</tr>
</tbody>
</table>

Table 8. Treatments incorporated into Woodstock across regimes

(SHEL70) must undergo a light shelterwood harvest (SHEL70) first. We also allow the model the option of not performing the next treatment in the sequence, subject to the constraints and the objective function. Woodstock works with stand-level data rather than tree-level data so we are not able to recreate the 'COVR' scenario based on the DBH threshold of individual trees as described in Table 3. Taking into account Woodstock’s chronological treatment sequence, we assume COVR is a reoccurring light shelterwood harvest removal (SHEL70), subject to the same area constraints in Table 8.
When designing constraints for Model II, the approach is nearly the same as Model I with one exception regarding treatment area. Rather than constraining the upper-bound of treatments based on total shelterwood harvest removals, we divide out initial-stage (SHELT40 and SHELT60) and final-stage shelterwood harvests (SHELT70):

\[ 0 \text{ acres} \leq H_t(SHELT40) \geq 1,410,000 \text{ acres} \]

\[ 0 \text{ acres} \leq H_t(SHELT60) \geq 1,410,000 \text{ acres} \]

\[ 0 \text{ acres} \leq H_t(SHELT70) \geq 1,785,000 \text{ acres} \]

For both Model I and Model II approaches, we assume the forest is capable of growing and harvesting wood eternally, and that markets are capable of in taking forest products (Gunn, 2007). These models are divisible, non-negative and deterministic - we assume that all relationships are understood perfectly and that the outcomes every action is known with certainty. Furthermore, we assume that site capabilities go unaffected by the models chosen management strategy, there are no changes in patterns of disturbance or climate, and that sampling weights from the FIA inventory represent aggregate forest cover characteristics. Moreover, the accuracy of LP models is highly dependent on the accuracy of growth and yield data. We keep these assumptions and limitations in mind when interpreting our results.
3 RESULTS

In this section, we review the results of our literature assessment, dividing out our initial calculations of net CO$_2$ sequestration before applying our findings to estimate the total statewide CO$_2$ sequestration potential across different management practices. Next, we overview our 100-year projections across 6 management regimes for FVS and calculate the total NPV across carbon and harvested timber revenue streams. Lastly, we present projections in forest growth and yield across management regimes from both Model I and Model II using Woodstock. Following the results in section 4.1, we compare and contrast total carbon stocks, carbon flux, and NPV across the three different methodologies, and recommend next steps for further research.

3.1 Literature Analysis Results

For the literature analysis, we collected over 100 observations from 9 studies. We calculate the mean, median, and variance of each category’s CO$_2$e flux rate above the study-specific baseline, which are summarized by management practice category in Figure 5. Detailed carbon sequestration and flux values by category are in Table 1 of the appendix. Changes in forest carbon fluxes, when compared to their study-specific baseline, vary widely within each management practice category. We represent sequestration as a negative rate and fluxes as a positive rate. As shown in the figure below, four practices contain both positive and negative fluxes and three have positive mean CO$_2$e flux rate for even-age management (0.10 tCO$_2$e/ac/yr), uneven-age management (0.08 tCO$_2$/ac/yr), and clearcut harvests followed by natural regeneration (0.04 tCO$_2$e/ac/yr).

Mean sequestration rates were highest for afforestation (-0.62 tCO$_2$e/ac/yr). This practice however, is subject to wide variation as displayed in Figure 1.4. Afforesting 240,000 acres of pastureland to forestland in Maine was found to have a mean additional flux of -1.84 tCO$_2$e/ac/yr compared to afforesting 6,000 acres of stream buffers at -0.05 tCO$_2$e/ac/yr (Cook-Patton et al., 2020). Consequently, the land-cover scale of which afforestation occurs impacts the rate of net sequestration within a study. Furthermore,
changes in study projections alter annualized sequestration rates, even when conducting the same practice. For example, when converting poor stocking levels to full stocking levels in Maine, rates vary between -1.38 tCO$_2$e/ac/yr, -0.69 tCO$_2$e/ac/yr and 0.24 tCO$_2$e/ac/yr over 30, 60, and 90 years respectively (Hoover and Heath, 2011). Flux rates standardized across a longer time horizon have lower CO$_2$e flux rates than rates standardized across a much shorter time horizon, attributable to fast forest growth dynamics in early versus late stages (Daigneault et al., 2021).

Incongruity of net flux rates within the even-age management category is attributable to differences in harvest targets, but more so from differences in study-specific baselines (see Table 1 for baseline assumptions). Conducting a shelterwood harvest (targeting a residual BA of 40 ft$^2$/ac every 50 years) is compared to a 'business as usual' scenario (flux of 1 tCO$_2$e/ac/yr) and yields a net CO$_2$e flux rate of 0.68 tCO$_2$e/ac/yr (Gunn and Buchholz, 2018). In contrast, an irregular shelterwood harvest (targeting a residual BA of 50 ft$^2$/ac every 80 years) compared to a 'high grading’ baseline (flux of 0.76 tCO$_2$e/ac/year)
results in a net sequestration rate of -0.45 tCO$_2$/ac/yr (Russell-Roy et al., 2014). Another dimension of variability between net flux rates stems from assumptions regarding carbon stored in HWP. To illustrate, a clearcut harvest (followed by natural regeneration), corresponds with a sequestration rate of -0.29 tCO$_2$/ac/yr over the length of the harvest including carbon stored in HWP (Russell-Roy et al., 2014). This rate is nearly the same from Nunnery and Keeton’s (2009) findings under the same 80-year clearcut scenario and HWP assumption (-0.28 tCO$_2$/ac/yr), but becomes a positive flux when standardized annually over 160 years (0.19 tCO$_2$/ac/yr) (Nunery and Keeton, 2010). The variation within one specific management practice alone is a result of differences in landscape scale, projection timelines, methodologies, and baseline assumptions. The appendix shows flux rates and difference in baseline rates. This literature review is limited in its ability to make a direct comparisons between the ‘additionally’ of CO$_2$e carbon flux rates between management practices.

Using the mean annual CO$_2$e net flux rate, and the average break-even carbon price, we calculate the total cost of mitigation and the total CO$_2$e flux in one year across the six categories, displayed in Table 9. We apply the average number of acres treated between 2009 and 2019 for avoided conversion, afforestation, clearcut, planting, even-age and uneven-age management regimes using the state Silvicultural Activity Report and FIA data (MFS, 2015). We attribute acres to the no-management category by standardizing 20% of the forestland (3.4 million acres) across 30 years. To simplify this analysis, we assume 16 million ‘unique’ acres of forestland are managed every 30 years across the state.
<table>
<thead>
<tr>
<th>Forest Carbon Practice</th>
<th>Acres Treated in Baseline</th>
<th>Mean Annual Net Flux (tCO$_2$ec/ac/yr)</th>
<th>Break-even Cost ($/tCO$_2$ec)</th>
<th>Total Net Flux (tCO$_2$ec/yr)</th>
<th>Total Mitigation Cost ($/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Afforestation</td>
<td>4,000</td>
<td>-0.62</td>
<td>$34</td>
<td>(2,480)</td>
<td>$84,320</td>
</tr>
<tr>
<td>Clearcut + Natural Regen</td>
<td>17,000</td>
<td>0.04</td>
<td>$17</td>
<td>680</td>
<td>$-</td>
</tr>
<tr>
<td>Clearcut + Artificial Regen</td>
<td>6,000</td>
<td>-0.32</td>
<td>$13</td>
<td>(1,920)</td>
<td>$24,960</td>
</tr>
<tr>
<td>Increase Stocking</td>
<td>-</td>
<td>-0.27</td>
<td>$17</td>
<td>(102,600)</td>
<td>$1,785,240</td>
</tr>
<tr>
<td>Extend Rotations (50 to 85 yrs)</td>
<td>-</td>
<td>-0.42</td>
<td>$19</td>
<td>(159,600)</td>
<td>$3,086,664</td>
</tr>
<tr>
<td>Uneven-age Management</td>
<td>185,000</td>
<td>0.08</td>
<td>$16</td>
<td>30,400</td>
<td>$-</td>
</tr>
<tr>
<td>Even-age Management</td>
<td>185,000</td>
<td>0.10</td>
<td>$12</td>
<td>38,000</td>
<td>$-</td>
</tr>
<tr>
<td>Pre Commercial Thinnings</td>
<td>10,000</td>
<td>-0.49</td>
<td>$14</td>
<td>(186,200)</td>
<td>$2,606,800</td>
</tr>
<tr>
<td>No Management</td>
<td>112,200</td>
<td>-0.44</td>
<td>$14</td>
<td>(1,496,000)</td>
<td>$20,944,000</td>
</tr>
<tr>
<td>Avoided Conversion</td>
<td>10,000</td>
<td>-212.00</td>
<td>$17</td>
<td>(2,120,000)</td>
<td>$36,040,000</td>
</tr>
</tbody>
</table>

Table 9. Average break-even carbon prices and net CO$_2$ flux rates across 10 forest carbon practice categories

Statewide Carbon Sequestration Potential

We find that the rate of 'additional' CO$_2$e sequestration from baseline forest management practices across the state is nominal, occurring at a linear rate of 24,638 tCO$_2$e annually. Under this baseline scenario, even and uneven age management practices cancel out 67% of mitigation achieved through no forest management, meaning a large share of net sequestration is attributable to gains from passive management. To stay consistent with out baseline logic, we did not include gains in sequestration from avoided conversion in our baseline determination, as that category implies a deviation in land use. Our baseline applies a ceiling for sequestration accounted for under the 'no management scenario' at 112,000 acres per year, representing the annual amount of passive forest management over 30 years. Next, we evaluate how changes in landowner behavior (switching from the baseline scenario to the increased mitigation scenario) change statewide forest sequestration overtime.

We re-distribute the implementation of 530,000 managed acres yearly by swapping carbon-emitting practices for carbon sequestering practices over the four scenarios. The net mitigation potential over 100 years as a share of annual mitigation by management...
practices, assuming a 100% participation rate, is shown in Figure 6. The mitigation potential of each scenario across 10%, 50% and 100% of landowner participation is shown in Figure 7.

**Figure 6. Cumulative mitigation in MMtCO2e per year across forest practices at a 100% participation rate**

The highest mitigation potential occurred for the extended rotations scenario 4 (increase the amount of extended rotations) with the highest average break-even carbon price of $16.23 tCO2e and total additional sequestration summing to 16.6 MMtCO2e under a 100% participation rate. High break-even carbon prices reflect the opportunity costs of delayed harvesting. However when the participation rate is cut in half, the net mitigation potential is also halved, totaling 8.3 MMtCO2e. Scenario 3 (increase rates of afforestation by 4) generates less than one-fifth of the total net mitigation potential under scenario 4 with an average break-even price of $13.81 tCO2e. Scenario 1 (clearcut and plant) has the lowest average break-even price of $13.48 tCO2e with a net mitigation potential of 13 MMtCO2e under a 100% participation rate.
Figure 7. Cumulative mitigation in MMtCO2e across across a 10%, 50%, and 100% participation rate

The increased stocking scenario (scenario 2) has a net mitigation potential of 9.6 MMtCO2e with an average break-even price of $14.55. Mitigation from scenario 4 makes up 67% of aggregate mitigation, compared to 65% for scenario 1, 58%, for increasing stocking (scenario 2), and 39% for scenario 3. We also tested the maximum mitigation potential in which all acres eligible for harvest annually would undergo the most rigorous individual treatments of clearcut and planting along with extending rotations. Cumulative sequestration over 100 years (assuming an implementation rate of 100%) was equivalent to 18 MMtCO2e for clearcut harvests followed by artificial regeneration, and 22 MMtCO2e for extended rotations. The total cost of mitigation was about $2.5 million dollars higher when only conducting extended rotations ($9.71 million) compared to the clearcut and plant scenario ($7.15 million).

We tested the sensitivity of our analysis by changing the amount of acres treated annually +/- 100,000 acres, totaling 13 million and 19 million uniquely managed acres over
30 years. Applied to the scenario with the lowest total additional mitigation potential (scenario 3), mitigation in year 100 (assuming a 100% participation rate) ranges from 2.7 - 4 MMtCO$_2$e. and between 13.5 - 19.7 MMtCO$_2$e) for the extended rotation scenario 4. Our analysis suggests that varying the number of acres treated across the landscape in addition to varying participation rates, can significantly alter total net mitigation from the forest ecosystem.

Our analysis makes many assumptions, the largest of which regards a constant rate of CO$_2$e sequestration throughout the next century. Because forests grow at a non-linear rate, using a constant rate of sequestration (which mirrors growth rates) in this analysis forms the lower and upper-bounds of total statewide net sequestration. This analysis is limited as we do not consider impacts of sequestration relative to statewide harvesting rates or forest composition. Additional sequestration is a relative measure based on a predetermined baseline, and different methodologies and studies define baselines differently. Trends in forest growth are highly dependent on the biophysical, social and political components. This analysis captures a wide range of additional sequestration potential in year 100 that can serve as upper and lower bound to be compared with 100-year net sequestration in our Woodstock model.

3.2 FVS: Determining NPV Across 6 Regimes

Across 17,000,000 acres, cumulative harvest removals are graphed across each of the 6 management regimes in Figure 8. BAU is the most harvest intensive by total volume (166 billion ft$^3$), followed closely by PCCT (154 billion ft$^3$) and CTH (146 billion ft$^3$). Rates of harvesting in the CTH regime nearly doubles that of the BAU regime after 20 years, while PCCT regimes outpace CTH harvests in 2074 as more stands undergo the last round of overstory removals, with periods of harvest recovery after 2079. As expected, COVR shows a rather steady rate of harvesting, averaging 4 billion ft$^3$ over 100 years. CCPL management regime increase immensely in the middle of the century as many stands
suddenly reach an average stand age of 50 between 2064 and 2084 but slows as most of the landscape is in the early phases of regeneration. Note that there are no external constraints to limit actions triggered by the simulation model, only biophysical constraints, which is why rates of harvest are highly exponential in the middle of the century.

Figure 8. Total cumulative harvest removals over 100 years

Figure 9 displays changes in the average AGC per acre over 100 years. This display aggregates growth and yield across all unique combinations of forest type, ownership region. Average live aboveground stocking in Maine can vary between 55 tCO$_2$e/ac for poorly stocked stands to 95 tCO$_2$e/ac for fully stocked stands. However, our FVS output projects such a rapid rate of growth, our stocking numbers surpass what we expect to be on the landscape after the first 10 years. The average CO$_2$ stocking per acre nearly doubles to 143 tCO$_2$e/ac over 20 years under the BASE regime, surpassing the biological maximum. We attribute this high rate of growth to a weak mortality function within the Acadian variant of FVS.
The total statewide NPV across 6 management regimes from 2019-2119 are evaluated across low, medium, and high carbon price scenarios using a discount rate of 5%. Using the calculations detailed in section 2.2, we estimate the NPV of each regime and apply a management cost $E$ of conducting pre-commercial thinnings and planting white spruce in the PCCT and CCPL regimes. This discounted total revenue from harvests alone is displayed first in Figure 10 where total NPV ranges from $0$ when no harvests occur under the BASE scenario, to $9.7$ billion under the CTH (commercial-thinning) regime. When factoring in the discounted cost of conducted pre-commercial thinnings, the PCCT regime is $195$ million less than CTH.

Figures 11 displays total NPV across the 3 carbon price programs and divides the present value from 3 sources: carbon flux, carbon stored in HWP, and harvested merchantable timber. The BASE scenario does not include harvest regimes and therefore only receives revenue from carbon credits. Only CTH receives a carbon price penalty for each tCO$_2$e that drops below the initial stocking conditions, which equal 1.27 billion tCO$_2$e.

Figure 9. Standing AGC stocking per acre per year

[Graph showing average aboveground CO2 stocking by management regime from 2020 to 2119, with distinct lines for each regime: BASE, BAU, CTH, PCCT, COVR, and CCPL. The graph has a y-axis labeled tCO2e/ha and an x-axis labeled 2019 - 2119.]

The graph illustrates the variation in average aboveground CO2 stocking across different management regimes from 2019 to 2119. The regimes include BASE, BAU, CTH, PCCT, COVR, and CCPL. Each line on the graph represents the stocking per acre per year for one of these management regimes.
Figure 10. Total NPV from timber harvests using a 5% discount rate stored in live aboveground timber in 2019. Revenues from harvests are constant throughout carbon price scenarios but diminish in proportion to other revenue streams from net gains in sequestration.
Figure 11. NPV across revenue streams under low, medium, and high carbon prices
When factoring in revenues and penalties from sequestration, overall trends under the $7, $20, and $51 carbon price scenarios reveal net emissions from the CTH regime. A carbon penalty between $7 - $51 is not enough to acquire net losses to any regime under any carbon price scenario. Despite profitable harvesting within the CTH regime as shown in Figure 10, CTH becomes less profitable with the addition of a carbon price with a $643 million, $1.84 billion and $4.68 billion dollar loss under RGGI, ARB, and SCC prices (Figure 11). Likewise, the regime with the highest net return is the BASE regime under ARB and SCC prices ($28 billion and $71.4 billion), but switches to the PCCT regime under the RGGI carbon price scenario, with a NPV of $11.3 billion after accounting for operational costs. Considering carbon stored in HWP, PCCT yields far greater returns, with saw logs comprising 56% of total harvests, compared to 19% for CTH and 23% for BAU. What could have been a slight penalty for PCCT is instead returned as a carbon credit.

Lowest returns for the RGGI carbon price program occur under a statewide implementation of the CCPL regime when including a $550/acre cost for artificial regeneration and delayed returns form harvests due to discounting. Presented in another way, 76% of the revenue earned from harvesting under CCPL are not realized until 2069. Under a $20 carbon price, the BAU regime has a NPV that is $55 million more than the PCCT regime due to higher gains in carbon sequestration. The less intensive COVR scenario has the third highest NPV under the RGGI carbon price, but becomes the second highest price under ARB and SCC prices.

Changes in the discount rate effect the net present value of both harvests and carbon credits; therefore, we test for changes in NPV under a 3% and 7% discount rate. Discounting does not affect the ranking of NPV across the 6 regimes under a $0 carbon price or a $51 carbon price, however prices deviate as increases in the discount rate are inverse with total NPV under RGGI and ARB programs. For example, under a 3% discount rate and $7 carbon price, the NPV of COVR is 2% higher than the BASE regime.
Table 10. Comparison of yearly sequestration and flux rates for 6 regimes statewide

<table>
<thead>
<tr>
<th></th>
<th>BASE</th>
<th>BAU</th>
<th>CTH</th>
<th>PCCT</th>
<th>COVR</th>
<th>CCPL</th>
</tr>
</thead>
<tbody>
<tr>
<td>100-year avg. net flux</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(tCO$_{2}$e/ac/yr) compared</td>
<td>-2.93</td>
<td>0.00</td>
<td>3.00</td>
<td>-0.09</td>
<td>-1.39</td>
<td>-0.85</td>
</tr>
<tr>
<td>to baseline</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>100-year avg. net flux</td>
<td>-3.45</td>
<td>-0.37</td>
<td>0.06</td>
<td>-0.46</td>
<td>-1.77</td>
<td>-1.23</td>
</tr>
<tr>
<td>(tCO$_{2}$e/ac/yr)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Under a 7% discount rate under the same carbon price, the NPV of the CTH regime is 3% more than the BASE regime as early returns on harvests are weighted more than later penalties for carbon lost to the atmosphere. Similarly under a $20 carbon price, CCPL has a total NPV that is $1.7 billion less than CTH as delays on harvest revenue outweigh returns from carbon sequestration.

When determining the additional mitigation potential over 100 years, we set the BAU regime as the baseline as it represents the 'business as usual' practice across the state, displayed in Table 10. As shown in the literature analysis, standardizing sequestration rates to an average across 100 years cannot capture short and long-term gains and losses in value, as exemplified by the CCPL regime. However, we annualized these rates to make comparisons across scenarios and methods. The BASE and COVR regimes have higher than expected rates of net flux compared to the baseline, with an average of 2.93 CO2/ac/year and 1.39 CO2/ac/year (Table 10). When compared to our literature analysis, these rates extend beyond upward bounds of net sequestration rates, due to a high growth rate and a weak mortality function within our FVS simulations. The 100 year average net flux rate from the PCCT regime (-0.46 tCO$_{2}$e/ac/yr) matches the mean additional sequestration rate from the pre-commercial thin treatment in the literature analysis (-0.49 tCO$_{2}$e/ac/year), as does CTH when compared with partial harvesting treatments (0.06 tCO$_{2}$e/ac/yr).
Determining total NPV across 6 management scenarios, 3 carbon price programs and 3 different discount rates reflects the trade-offs stakeholders face when conducting long-term forest management planning. Not reflected in this method is the reality that the Maine forest is managed in many different ways for both carbon-sequestering and harvest-intensive purposes. It is unlikely that the entire landscape will adopt one regime based on the biophysical conditions of the forest and according to policy and socioeconomic constraints. Our next section displays our results of adding socioeconomic dimensions using forest modeling to determine the suite of regimes that maximize total carbon stocking, and net present value.

3.3 Scenario Modeling with Woodstock

Using the linear program solver, we allow Woodstock to determine one of the six regimes pathways for each plot that maximizes the objective function. Recall from section 2.3 that there are 8 objective functions, however we will focus on disseminating the results from the first 5 scenarios: (1b) establishing a historical baseline, (#2) maximizing total carbon stocks, (#3) maximize total harvest volume, (#4) maximize total net revenue, and (#5) maximize total NPV from harvest revenues. Each of the objective functions are presented with harvest constraints only (a) or harvest and area constraints combined (b) detailed in section 2.3, in 5-year increments from 2019 to 2119. We analyze the trade-offs of each scenario, relative to their maximization objective and constraints. Note that we analyze key scenarios in this section and place all other detailed results in the appendix.

Model I: Regimes Based Approach Results

In Figure 12, we present the models allocation of regimes in acres across 4 objective functions with no constraints, compared to the historical baseline scenario which reflects harvest and area constraints. When maximizing total CO$_2$e stocks in scenario 2, the model predominately chooses the BASE (no management) regime to capture high growth rates that lead to higher AGC stocking levels than the initial stand conditions in 2019.
Figure 12. Model I: Total acres across 6 management regimes for 5 unconstrained maximization functions

Interestingly under scenario 2, the model chooses to continuously thin the stand (COVR) for 22,000 acres, clearcut and plant (CCPL) for 6,000 acres, and pre-commercial thin (followed by a shelterwood and OSR) for 16,000 acres, as stocking conditions in 2119 for certain plots are higher than in 2019. When maximizing for harvest volume, the model prefers to select the BAU regimes as opposed to optimizing for maximum revenue, in which highly merchantable trees are cut via the PCCT regime. When maximizing for NPV under a 5% rate, the model chooses the pathway that harvests earlier in the modeling time frame.

When the scenarios are constrained to reflect statewide harvest and treatment area bounds, the allocation of regimes changes significantly. The model is constrained such that more than half of the landscape is allocated to no management (BASE). Area and harvesting constraints combined (#.b) have more acres devoted to the BASE scenario than harvest constraints alone (#.a) across all scenario groups as shown in Figure 13. Logically, the less acres that qualify for treatment in each time period, the distribution of
management practices is altered to satisfy the objective relative to the constraints. For example, in scenarios 3a, 4a and 5a, the model employs the COVR regime equally if not more than the CTH regime. However, when adding area and harvest constraints (b), acres allotted to no management increase 8% and 15% with a dramatic reduction in the continuous cover (COVR) regime, compensated by increasing the proportion of acres under the business-as-usual shelterwood regime (BAU). Previously in Table 7, we denote that the maximum amount of acres in shelterwood harvests (BAU) is 13 times more than area allocated for the estimated amount of acres allotted for continuous cover (defined as crop-tree-release cutting) (MFS, 2015). The model chooses the CTH and BAU regimes instead of the COVR regime in order to reach the objective function. This same trend is observed for the clearcut and plant regimes (CCPL).

Total harvests over 100 years in scenarios 3b and 5b are 1% less than total harvests under scenarios 3a and 5a. Between (.a) and (.b) scenarios, the objective function’s
solution average a 1% difference in total CO2 stocking, and a 5% difference in total net revenue and total net present value. In other words, the final objective is largely attributable to harvest constraints from scenario (#.a) rather than area constraints from scenario (#.b) despite having different allocations of treatments. As both harvest and treatment area constraints reflect historical realities, moving forward, we present only the results from scenarios (#.b). Figure 14 displays total harvest volume by management regime. Total harvest volumes for unconstrained scenarios and scenarios with harvest constraints only (a) are displayed in the appendix.

![Figure 14. Model I: Total harvest volume by regime](image)

Intensive harvesting is disincentivised in order to maximize the current forest stocking and carbon stored in HWP. Therefore, total harvest volumes equal the low harvest bound of 2.9 billion ft³ in each period for maximizing CO2 stocks (scenario 2b) and equal the upper bound of 3.3 billion ft³ when maximizing harvest volumes (scenario 4b). The distribution of management regimes as a total of harvest volume in Figure 14 mirror the distribution under treatment area in Figure 13. However the chart displays a greater
proportion of harvest volume occurring from PCCT and CTH management regimes as opposed to COVR and CCPL. Under profit-maximization scenarios 1b, 3b, and 5b, the proportion of sawtimber relative to pulpwood is 46%, 47% and 44% respectively compared to only 34% when maximizing harvest volume only in scenario 4b.

Figure 15 illustrates the total AGC in tCO$_2$e/acre from 2019-2119. Constrained scenarios (shown via dotted lines) are higher than unconstrained profit and harvest unconstrained scenarios with an average stocking of 293 to 350 tCO$_2$e/acre in 2119. Scenarios 3 and 4 mirror the statewide application of BAU and PCCT regimes from Figure 9, while scenario 2 mirrors the BASE regime.

When optimizing NPV (at a 5% discount rate) across different carbon prices (under harvest and area constraints), we add a constraint in which AGC stocking cannot decline between periods. In this way, landowners in this scenario do not have the opportunity to pay a penalty for each tCO$_2$e lost between periods if they choose to harvest, reflecting the
'additionality' of carbon offset projects. We assume constant carbon prices, consistent with our constant stumpage prices. The model selects the same distribution of pathways across scenarios 6b, 7b, and 8b, resulting in the same total harvest volume in 2119. In other words, the model does not optimize different outcomes when considering different carbon prices. This is likely due to stringent constraints on total harvest, treatment area, and positive gains in AGC. When CO2 stocks are optimized without constraints, the model selects only the BASE regime for the 3 scenarios, capturing the fast growth of the BASE regime while gaining credits on every ton sequestered above the previous period.

When changing the discount rate from 5% to 3% and 7%, changes in the allocation of regimes are negligible as the model is heavily constrained by non-declining AGC. As shown in Figure 16, total NPV increases as carbon price increases and declines with respect to the discount rate. Under a 5% discount rate, total NPV is $5 billion for scenario 1b but increases to $6 billion when maximizing NPV with no carbon price (scenario 5b). This figure doubles to $13 billion when considering rewards from sequestration under a $7 carbon price and a 5% discount rate. Likewise, total NPV increases to $26 billion under a $20 carbon price and $58 billion under a $51 carbon price. Variations in the discount rate can affect total NPV; however because the model selects the same treatments for the same plots regardless of the carbon price scenario, no new trade offs can be made to receive a better return under a different discount rate.

To compare carbon sequestration rates, we take the difference between initial AGC stocking and final AGC stocking in year 2119. We then subtract total sequestration over 100 years for each scenario from the historical baseline scenario (1b). Note that carbon flux rates are representative of AGC and HWP pools. Table 11 displays the total CO2 flux above the baseline, annual sequestration (in tCO2e/acre), and the total net revenue across each practice. Scenario 2b has the most additional mitigation potential over 100 years, storing an additional 11 MMtCO2e when harvesting on the lower bound and incorporating high cost and high growth practices like CCPL and PCCT regimes. Scenario 5b follows by
Figure 16. Model I: Total NPV across 3%, 5%, and 7% discount rate for RGGI, ARB, and SCC carbon prices

sequestering an additional 4.5 MMtCO$_2$e over 100 years by employing intensive CTH and PCCT regimes.

Despite a similar regime application, scenario 3b has the same sequestration potential as the historical baseline (1b) but a slightly higher net revenue as the model selects a different composition of stands to harvest with more merchantability than scenario 1b as shown in Table 11. Scenarios 6b, 7b, and 8b have a similar regime composition and carbon sequestration rate (tCO$_2$e/acre/year) as 1b, but slightly less net revenue as the model selects to retain wood with higher merchantability. The non-declining AGC stocking constraint means that stocking never drops throughout 100 years, but total AGC stocking in 2119 is slightly less than total stocking under the historical baseline scenario. Due to the design of Model I in addition to stringent harvest and area constraints, total net revenue is comparable across all scenarios (between $24 and $26 billion) with the exception of
scenario 2b which harvests low grade lumber under high-cost regimes (PCCT and CCPL), and therefore about 40 percent less than the other scenarios.

One limitation with this approach is that it only accounts for net losses in CO2 from harvests, not from dead or decaying wood nor from disturbances such as fire, pest or disease. Due to the weak mortality function within the FVS BASE curve, the annual net forest carbon sequestration rate is higher than we would expect. Model I projects an average rate of 37 MMtCO₂e/year under the historical baseline scenario, compared to an average rate of 8-10 MMtCO₂e/year (Bai et al., 2017; Domke et al., 2021).

### Table 11. Model I: Total CO2 flux, flux compared to the baseline, and total net revenue across constrained scenarios. Note that NPV is calculated via a 5% discount rate

<table>
<thead>
<tr>
<th>Model I</th>
<th>1b. Historical Baseline</th>
<th>2b. Max CO2 Stocks</th>
<th>3b. Max Net Revenue from Harvests</th>
<th>5b. Max NPV</th>
<th>6b. Max NPV RGGI ($7)</th>
<th>7b. Max NPV ARB ($20)</th>
<th>8b. Max NPV SCC ($51)</th>
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<td>Additional CO2 Flux From Baseline (MMtCO₂e/yr)</td>
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<td>(2.84)</td>
<td>(-2.19)</td>
<td>(-2.46)</td>
<td>(-2.18)</td>
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<td>Net Revenue from Harvests (billion $)</td>
<td>$24</td>
<td>$15</td>
<td>$26</td>
<td>$24</td>
<td>$23</td>
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<td>$23</td>
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<tr>
<td>Net Revenue from Harvests &amp; Carbon Credits (billion $)</td>
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<td></td>
<td>$50</td>
<td>$98</td>
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Model II: Treatment Based Approach Results

By adding another model dimension of time in which treatment $m$ differs across a single rotation period $t$ across all stands, we model the relationship between optimal harvest timing differently, subject to each unique maximization function, relative to its constraints. Like Model I, there are negligible differences occur between harvest-only constraints (#.a) and harvest and area combined constraints (#.b) scenarios. In this section, we focus on the comparing the constrained scenarios and the historical baseline (1b). Unconstrained model results (in addition to scenarios results) are displayed in the appendix.

The largest allocation of ‘no management’ (or the BASE regime) is present when maximizing total CO2 stocks (scenario 2b) and maximizing harvest volume (scenario 4b) in Figure 17. However, the model produces unreliable results when optimizing for maximum
Figure 17. Model II: Total acres across 6 management regimes for 5 constrained maximization functions

harvest volume while also constraining total harvest volume, such that it relies more heavily on area constraints. The model cuts more intensively on fewer acres than in other scenarios (see Figure 18). A second observation is that PCCT is not allocated under any constrained scenarios, as the action is not an efficient means of maximizing revenue (or harvest volume) relative to its implementation cost. This is also likely occurring because conducting pre-commercial thinnings in itself changes the response of the growth curve substantially, which we do not account for in our aggregate yields curves we input into Model II (see Figure 4). When maximizing harvested wood product and AGC stocking (scenario 2b), the model employs the clearcut and plant regime in order to meet harvest objectives (staying in the lower bound of 2.9 billion ft² harvested in each period), while increasing sequestration. Profit-maximization objective functions 1b and 3b look nearly identical, with both using CTH regimes to collect high net returns on stands with high merchantability before employing lighter shelterwood harvests with the COVR regime or one shelterwood harvest followed by an OSR in the BAU regime. Unlike Model I, profit-maximizing scenarios vary widely across harvested timber grade, with sawtimber comprising 43% of total harvests in the historical baseline and maximizing CO2 stocks (scenario 1b and 3b), 45% when

51
maximizing total harvest volume (scenario 4b) and 30% when conducting early commercial thin and shelterwood harvests when maximizing NPV constrained (scenario 5b).

Figure 18. Model II: Total harvest volume by regimes

Because Model II optimizes harvest timing in addition to treatments, we demonstrate harvest intensity overtime in Figure 19. When optimizing for NPV under a 5% discount rate unconstrained (scenario 5), the model conducts harvests above historical baseline before slowing the rate of harvesting in 2064. When optimizing net revenue unconstrained (scenario 3) the model delays harvests until stands are able to produce large sawlogs at the turn of the century, compared to scenario 4 which starts intensive harvesting 15 years prior as growth rates begin to slow. Scenario 2 tracks along the x-axis with very few removals over the century. Constrained scenarios follow constant harvest trajectory with scenarios 4b and 5b slightly higher than the baseline (as shown in grey).

Figure 20 is the opposite of Figure 19, displaying average tCO₂e stocking per across each scenario. As expected, the lowest AGC stocking at the end of the 100-year horizon corresponds with unconstrained harvest and revenue maximization functions 3 and 4 with 39 tCO₂e/acre and 24 CO2e/acre respectively. Interestingly, maximizing NPV unconstrained is the only scenario that has a later period of forest growth and recovery in
Figure 19. Model II: Total harvests for scenarios 1.b - 5.b

Figure 20. Model II: Total AGC stocking per acre for scenarios 1.b - 5.b
which total stocking in the unconstrained scenario bypasses scenario 5b in 2104.

Unconstrained scenarios that maximize CO2 stocking have the highest standing AGC per acre with 417 tCO$_2$e/acre in 2119, and 295 tCO$_2$e/ac when constrained (scenario 2b).

Figure 21. Model II: Total treatment area allocation for scenarios 1.b, 6.b - 8.b

When optimizing NPV (at a 5% discount rate) across different carbon prices (under harvest and area constraints), we add a the non-declining AGC constraint as we did in Model I such that no penalties are issued from harvesting emissions. We assume constant carbon prices, consistent with our constant stumpage prices. Unlike Model I, Model II selects slightly different treatments to occur on different stands and employs intensive harvesting treatments. As displayed in Figure 21, the model selects a combination of commercial thinning and shelterwood harvesting that make up the CTH, COVR and BAU regimes when maximizing NPV. As carbon price increases, more area is allocated to the CCPL treatment as higher net returns offset costs of planting. For RGGI carbon prices under a 5% discount rate, 1.2 million acres are left untreated in the BASE scenario,
compared to 930,000 acres under a $20 carbon price and 570,000 under a $51 carbon price. Harvests occur more in the later periods as the scenario maintains a lower threshold of AGC stocking as the carbon price increases in order to meet the maximization function relative to the constraint (see Figure 22).

![Aboveground CO2e Stocking per Acre](chart.png)

**Figure 22.** Model II: Total AGC stocking for scenarios 1.b, 6.b - 8.b

We then test NPV by applying a 3% and 7% discount rate in Figure 23. Unlike Model I, total harvest volume, allocation of regimes, and net revenue shift across carbon price scenarios such that it would be more profitable to have a carbon price of $7 under a 3% discount rate than apply a $51 carbon price under a 7% discount rate. Harvest intensity is 1 billion higher for the RGGI scenario (6b) under a 3% discount rate. However when applying a 7% discount rate, the SCC scenario (8b) has a higher harvest intensity, removing 1-2 billion ft³ more. Total NPV for no carbon price being an average of 35% higher in Model II than in Model I for the historical baseline scenario. But when factoring in a non-declining AGC constraint subject to maximizing NPV in scenarios 7b and 8b,
total NPV is between 27% and 56% larger than Model I. Total NPV in Scenario 6b is on average 5% larger in Model II than Model I. Despite implementing a very similar suite of treatments, harvest timing varies according to both the carbon price and the discount rate.

Figure 23. Model II: Total NPV across 3%, 5%, and 7% discount rate for RGGI, ARB, and SCC carbon prices

Because Model II has more flexibility when registering treatments, it sequesters more carbon under carbon maximization function (#2), and emits much more under harvest volume, net revenue and NPV maximization functions. Like the literature analysis method, it was important to include a historic-informed baseline scenario for comparison purposes. Unconstrained scenarios project ‘additional’ mitigation potential to be 2-5 times the rate under harvest and area constraints. When constrained by minimum harvesting levels, fluxes range between 5 MMtCO$_2$e when maximizing harvest volumes (#4b), and sequester 2.8 MMtCO$_2$e when maximizing stored and sequestered CO2 (#2b) under Model I compared to the baseline. Model II conveys more positive fluxes (emissions) than negative
Table 12. Model II: Total CO2 flux, flux compared to the baseline, and total net revenue across constrained scenarios. Note that NPV is calculated via a 5% discount rate

We compare carbon sequestration rates from Model II in Table 12. The historical baseline scenario in Model II has an annual sequestration rate of 8.0 MMtCO$_2$e/year, falling within the realm of current statewide estimates (Bai et al., 2017; Domke et al., 2021). Because growth rates are higher than average under the BASE curve, Model II takes advantage of fast growth following harvests when maximizing NPV and total CO2 stocks. Compared to the baseline, scenarios 5b through 8b sequester an additional 2.9 - 5.5 MMtCO$_2$e above the baseline over 100 years. Under scenario 2b when total carbon stocks are maximized by means of an intensive CCPL regime, additional sequestration is upwards of 34 MMtCO$_2$e over 100 years.

When factoring in the cost of planting and the revenues from lower grade timber harvests, net revenue is the lowest across the six scenarios at $18 billion. Interestingly, net revenue from scenario 3b is $3 billion more than applying a RGGI price of $7 under a non-declining AGC constraint, but becomes $12 billion and $45 billion lower when adding a $20 and $51 carbon price. Similar to Model I, scenario 3b has a similar mitigation potential to the historic baseline as they both seek to maximize profits, despite a lower bound for harvesting under the baseline scenario. Average annual flux rates are much lower than Model II across most scenarios as 2119 AGC stocking levels are about half of 2119 levels in Model I.
4 DISCUSSION

In this thesis, we constructed a statewide forest sector model framework using Woodstock to determine the 100-year sequestration potential, CO2 stocking potential, mitigation costs and net present-revenues based on changes in forest management based on the current composition of Maine’s forestland. Illustrated in Figure 3, this framework uses two modeling methods (regimes based and treatment based approach), inputs Maine FIA forest plot data with simulated growth and yield curves (over 100 years) via 6 different management regimes, using FVS. We selected a no-management regime (BASE) and 5 other commonly applied regimes across the state with varying mitigation potential and costs. Because we use a forest vegetation simulation model (FVS) in addition to a linear program model (Woodstock), the results of our objective functions are directly linked to the strength of our growth and yield curves.

The accuracy of our ‘no management’ or BASE regime is questionable. The BASE regime has growth rates that sequester 5 times the historical statewide average under the Acadian variant of FVS due to a weakened mortality function (Daigneault et al., 2021; Domke et al., 2021). If forests were truly this productive, it would have major implications in statewide land use and market dynamics. Despite this, we still applied these FVS curves in the Woodstock model to keep the FVS simulation methodology consistent across 6 regimes. In Model I, the BASE regime serves as one of 6 ‘regime pathways’ by which the model can employ in each ‘plot’ to achieve its objective function. In Model II, the BASE curve serves as the basis of which informs the growth rates across 7 forest types following harvest treatments (Figure 4). The accuracy of growth and yield curves are important as it informs model trade-offs concerning when to harvest, what to harvest, and the best mode of harvesting. To improve the accuracy of our Woodstock models objective functions, a future iteration of the model should include growth rates under the BASE regime that reflect growth rates closer to historical trends. However the
current state of the Maine Integrated Forest Sector Model provides a strong foundational framework that can continue to be expanded upon and modified over time.

Before implementing the framework, we first examined current literature on forest carbon sequestration in the northeast, synthesized their results, and applied their findings to Maine’s 17 million acres of working forest land. We determine that under historical trends in land use, Maine can sequester an additional 3.4 - 16.2 MMtCO$_2$e. We also determined that the most cost-effective approach incorporates clearcut and planting treatments and that the approach with the highest mitigation potential applies extended rotations or delaying harvests an additional 20 to 25 years. However, our analysis is heavily limited on the assumption that carbon is sequestered at a constant rate over time, and based on the prior analysis regarding mean sequestration rates by practice. We make no assumptions regarding changes to harvesting rates or revenues from harvesting. To narrow this estimate and to incorporate statewide trends in forest growth and yield, we apply the framework and use FVS simulations across 6 regimes and incorporate maximization functions to compare trade-offs across maintaining carbon pools and harvest intensities.

In our FVS approach, we estimate the financial returns and carbon stocks of implementing various practices in FVS out 100 years, and determine that the PCCT (pre-commercial thin regime) yields the highest returns ($11.3$ billion) under a $7$ carbon price as carbon stored in HWP is rewarded and as gains in total AGC stocking outweigh losses from harvesting emissions. Total NPV increases to $28$ billion and $71.4$ billion under the BASE regime as the carbon price increases to $20$ and $51$ per tCO$_2$e. Mitigation potential is the strongest under the BASE regime ($3.45$ tCO$_2$e/ac/yr, or about 59 MMTCO$_2$e/yr in total due to high growth rates and a weak mortality function from the FVS simulations. Under this approach, we do not apply constraints for harvesting and treatment activity as we can in our final approach using Woodstock.

Our framework uses the Woodstock to model a regimes-based approach (Model I) and a treatment-based approach (Model II) to quantify potential supply impacts to the Maine
forest sector. We analyze several different scenarios with respect to historical harvesting and treatment constraints. Different maximization functions produce differing optimal solutions relative to the defined constraints. Overall trends under the Model I approach reveal that the best practices for increasing forest carbon in HWP and live AGC pools include a mix of the continuous cover regime (COVR), and the business as usual regime (BAU) as these regimes harvest the least intensely out of the 6 choices. When allowing the model to optimize treatment timing and integration under a treatment-based approach (Model II), the clear-cut and planting practice is used extensively to maximize total carbon stocking. Furthermore, under a Model II approach, more intense harvests can occur in later periods which decrease the total amount of AGC in the year 2119 relative to Model I which pre-determines forest recovery and growth following intensive harvests in the mid-century. We report that Model I can sequester an additional 0 - 11 MMtCO$_2$e above the baseline (scenario 1b) whereas Model II sequesters an additional 0 to 34 MMtCO$_2$e. Outside of maximizing total carbon stocking (scenario 2b), this range falls between 0 and 5.5 MMtCO$_2$e which is within the realm of our literature analysis estimates.

As a secondary research goal, this thesis attempts to analyze the differences and trade-offs between Model I and Model II approaches. The strength of the Model I approach compliments that of biophysical forest inventory modeling: each stand maintains its unique growth rate as treatments are applied. Additionally, the completion times for solving scenarios Model I are five to eight times faster than the total solve time of scenarios in Model II. The weakness of this approach concern the models lack of ability to optimize the use and occurrence of individual treatments for periods after the conditional threshold for harvesting is met. In reality, landowners are constrained by time, money, technical, and harvesting thresholds. This reality is integrated in Model II such that the model has greater flexibility to satisfy objectives and constraints, allowing the modeler to analyze a greater set of decisions.
However, we cannot form like-to-like comparisons between Model I and Model II, largely due to the way we structure their constraints. Regime area constraints in Model I reflects the total area undergoing any stage of shelterwood harvest activity as compared to treatment-specific constraints in Model II because we cannot parse out individual treatments and their occurrence under a regime-based approach. For example, a final stage shelterwood harvest can only occur on a maximum of 1.7 million acres in Model II as opposed to 3.18 million (under CTH and PCCT regimes which both include a final shelterwood harvest treatment) in Model I. Model II can chose optimal timing relative to the constraints more easily than it can choose across 6 pathways in Model I. Therefore, we see intense harvesting occurring on fewer acres in Model I than in Model II. In this way, the average AGC stocking in Model II is less than in Model I as more acres undergo some form of harvesting, following a lower trajectory of growth as opposed being left alone entirely under the BASE regime.

All in all, many different exogenous inputs, assumptions, and constraints impact the results of our Woodstock models. Both models reveal that conducting harvests on stands well-suited to fast growth and recovery can use a mix of intensive (OSR and clearcuts) and partial harvests (shelterwood, pre-commercial and commercial thinnings) can increase forest carbon stocking while also satisfying historical demand for harvested wood products over the long term. Fast-growing forest plots are suitable to conductive intensive harvests which can regenerate quickly, thereby sequestering more carbon. Selecting a different suite of management practices suitable to the unique conditions of each stand can maximize the carbon storage capacity and maintain constant harvest rates. Collectively, our multifaceted approach and analysis indicates that it is possible to implement a diverse of regimes that average a carbon sequestration rate of -0.85 tCO$_2$e/ac/year and produce HWP C storage of 3 MMtCO$_2$e/year or more can help Maine reach statewide carbon neutrality by 2045, assuming constant declining trends in gross GHG emissions as well.
When comparing the model results with what we find in the literature assessment, our mean annual carbon sequestration rates are high. For example, a 100-year no management practice sequesters -1.40 tCO$_2$/ac/year (Russell-Roy et al., 2014), compared to -3.45 tCO$_2$/ac/year (over 100 years) under the BASE regime. In the dominate literature, historic rates of net mean carbon sequestration in Maine are between -0.47 to -0.6 tCO$_2$/ac/year for the past 20 years (Domke et al., 2021). Our baseline scenarios have a net mean carbon sequestration rate between -2.19 tCO$_2$/ac/year in Model I and -0.47 tCO$_2$/ac/year in Model II. Model II baseline aligns with Domke et al., 2021 sequestration figures, but Model I harvests more intensively on fewer acres than in Model II due to constraining regimes as opposed to individual treatments, therefore sequestration rates on a plot level have much wider variation than in Model II.

Mirroring high growth rates, the total NPV figures (of timber harvesting only) are between 0 and 400 per acre (or between $3 and $7 billion), which is higher than the NPV estimates from a 100-year Maine projection by Wagner et al., 2010. Despite using similar harvest constraints, this discrepancy in total NPV between our study and the aforementioned is attributable to discount rate (6% vs 5%), our applied management regimes, our treatment area, and our exogenous stumpage prices which we differentiate by 9 regions and 7 species groups as opposed to 3 species groups (which may have a lower mean price). High growth rates in our model increases the proportion of saw timber (as opposed to pulpwood) which can dramatically increase the harvest value. Constraining the annual pulpwood and saw timber harvest could change the total NPV across scenarios.

4.1 Limitations and Suggestions for Future Research

This modeling framework provides a basic foundation for a statewide forest sector model which analyzes trade offs between increasing forest carbon storage while maximizing profits from harvesting. Our model is heavily dependent on our assumptions regarding harvesting bounds, treatment area, discount rate, and constant exogenous prices. Incorporating changing prices for timber and carbon could enhance the model’s long-term
forest planning ability. We do not adjust for inflation for stumpage prices, nor do we increase carbon prices overtime. We also assume forest area remains constant and that no forestland is lost (or gained) from changes in land use. We do not assume changes in forest composition (more acres from one forest type into another), nor changes in the rate of stand mortality outside of FVS.

Although linear program models incorporate shadow prices which reflect opportunity costs landowners face when managing forestland, it does not account for multiple objective functions at one time like non-linear programming. Many landowners across the state might be profit-maximizing or carbon-maximizing but have differing discount rates which may impact their decision to harvest. Our results reflect the potential sequestration, storage, and revenue generated across a mixed management approach. Actualized changes in forest management rely heavily on the participation of private landowners across the state which own 95% of forestland statewide (MFS, 2015). Private ownership in Maine is comprised of family owners (which make up 32% of forestland ownership), private investors (40%), forest products industry or commercial landowners (13%) and private conservation groups (6%), each which are more likely to employ certain management practices over others (USFS, 2021). Notably, small family landowners are more likely to employ some form of partial harvesting or no harvesting rather than clearcut and plant regime which is only performed in Maine by commercial landowners (Sass et al., n.d.). With this in mind, applying additional constraints on management action (or inaction) and annual harvesting intensity by ownership class would enhance model results.

Furthermore, we assume CO$_2$e storage as a factor of each forest type per ft$^3$. This is a broad assumption, as our carbon calculations occur at a stand-level as opposed to our growth and yield projections which occur at a tree level. In this way, increases in carbon storage are proportional to increases in stand volume as opposed to individual tree volume. The accuracy of live, AGC pools can be expanded upon by calculating CO$_2$e content at an individual-tree level using biometric calculations or by using the FVS Fire and Fuels
extensions (Crookston and Dixon, 2005). Additionally, carbon loss from dead or decaying wood could be integrated into the Woodstock model on a plot level. By integrating carbon storage across various aboveground pools, the model can capture the choice to harvest products before forest decay, or leave it in the forest to decay (relative to harvesting costs), having implications for maximizing carbon sequestration and storage.

Future adjustments can be made to improve the optimization model outside of improving the accuracy of our live, aboveground carbon storage calculations. The first suggestion is to extend the Woodstock model horizon from 100-years to 150 or 200 years when evaluating unconstrained scenarios. In this way, the model does not delay harvest action until the end of the century, as is common for dynamic forest LP models. In our current version of Model II, the PCT action does not illustrate sharp regrowth following the treatment along the aggregate growth curves by forest type, and therefore the model does not choose to integrate PCT in any constrained scenario. Secondly, including empirical growth curves for stands that meet low, medium, high and overstocked thresholds in addition to the growth response following a pre-commercial thin treatment would fine-tune the model to make better forest management allocations across the landscape. Third, we note that the growth and yield modeling from FVS are likely to be overly optimistic. Future work should refine these curves using alternative FVS parameterization with a stronger mortality function or another growth and yield model. This issue is of particular importance when looking over such a long time horizon, as we did for this analysis.

As this approach integrates both FVS simulations and linear programming, many more opportunities for research can be developed using this framework. As mentioned throughout this thesis, simulating forest growth and yield with a stronger mortality function could produce different model results. Creating scenarios in FVS which apply a different or broader suite of management practices than the treatments and regimes used in this analysis give the model a broader choice set from which to optimize. Additionally, with growing concerns regarding the resiliency and adaptation of Maine’s forests in the
future, FVS is able to impose certain environmental futures and events such as an emerald ash borer outbreaks, drought, or forest fires which can be integrated into a Woodstock model, allowing researchers to examine optimal management and trade-offs in the face of environmental shocks.

The above suggestions for future research regarding changing exogenous prices, ownership constraints, FVS simulations, time horizon and aboveground carbon pool metrics require multiple months and years of model tweaking, technical capabilities, and expertise. The above additions can enhance the models accuracy and reliability without compromising its simplicity or operation time. This thesis provides forest planners and researchers with a framework which can inform decisions regarding policy, landowner engagement, and management for the state of Maine.


Woodall, C. W., & Weiskittel, A. R. (2021). Relative density of united states forests has shifted to higher levels over last two decades with important implications for future dynamics [Number: 1 Publisher: Nature Publishing Group]. *Scientific Reports, 11*(1), 18848. https://doi.org/10.1038/s41598-021-98244-w


### APPENDIX

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<th>Max Net Flux (tCO2e/ac/yr)</th>
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<th>Mean Break-Even Price ($/tCO2e)</th>
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Table A.1. Carbon sequestration and flux summary statistics across 9 management practice categories
Figure A.1. Model I: Total harvest volumes across all combinations of scenarios and constraints by harvest regime
Figure A.2. Model II: Total treatment area across all combinations of scenarios and constraints by regime
Figure A.3. Model II: Total harvest volumes across all combinations of scenarios and constraints by harvest regime
BIOGRAPHY OF THE AUTHOR

Logan Woodyard descends from a long line of Appalachian loggers, foresters, and hardwood mill owners. Born in southeastern Ohio, Logan attended college at The Ohio State University as a first-generation college student where she received a full-ride scholarship to study Agribusiness and Applied Economics. During her time at Ohio State, she interned for the Ohio Department of Medicaid, assisting with caseload modeling, healthcare policy and budget projections. After receiving her B.S. from The Ohio State University, her passion concerning rural community resilience, interest in modeling, and family connection to the lumber industry led her to study forest modeling, economics and policy under Dr. Adam Daigneault through the University of Maine. Upon graduation, Logan will work as a acquisitions and natural climate solutions analyst through Green Diamond Resource Company out of Seattle, Washington. Logan Woodyard is a candidate for the Masters of Science degree in Forest Resources from the University of Maine in August 2022.