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Life-cycle impact assessment of hardwood forest resources in the eastern United States

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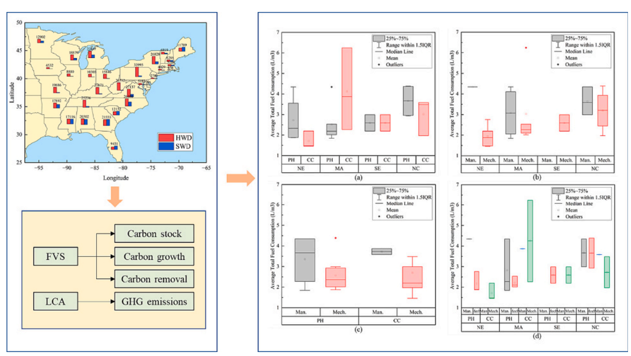
HIGHLIGHTS

- LCA was integrated with FVS to analyze the carbon sequestration potential of hardwood forests in the eastern United States.
- GHG emissions from manual and mechanized harvesting systems ranged from 9.13 to 12.15 kg CO_2e/m^3.
- Uneven-aged management resulted in a forest carbon balance of 7.72 kg/m^2 and a net carbon balance of 1.69 kg/m^2 for the 9.99×10^{11} m^2 hardwood forests in the eastern United States annually.
- Even-aged management yielded a carbon balance of 6.20 kg/m^2 and a net carbon balance of 0.91 kg/m^2 for the 9.99×10^{11} m^2 hardwood forests in the eastern United States annually.
- The Fire and Fuels Extension of FVS did not account for soil carbon, encompassing both dead tree roots and live tree fine roots.

GRAPHICAL ABSTRACT



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ABSTRACT

To explore the carbon sequestration potential of hardwood forests in the eastern United States, the forest vegetation simulator (FVS) and life cycle assessment (LCA) were integrated to analyze the forest carbon dynamics for the four subregions of the eastern United States: northeast (NE), mid-Atlantic (MA), southeast (SE), and north central (NC). This study quantitatively assessed current forest management practices for timber production and their associated life-cycle environmental impacts. The system boundary was selected to be consistent with the A1 module (extraction and upstream production) required by an Environmental Product Declaration (EPD) for wood products. The results indicate that uneven-aged (UA) forest management yields higher carbon stocks and growth than even-aged (EA) management across all subregions. In contrast, clearcutting under EA management results in higher carbon removal. It was found that fuel consumption-related greenhouse gas (GHG) emissions for manual

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and mechanized harvesting systems for both management types ranged between 9.13 and 12.15 kg of CO_2 equivalent per cubic meter (kg CO_2e/m^3), with an average of 11 kg CO_2e/m^3 of hardwood timber harvested across all subregions. It is estimated that 63–187 megajoules (MJ) of energy is needed to produce 1 m^3 of hardwood sawlogs. The extraction and loading processes contributed more to the total GHG emissions than the felling and processing within the system boundary. The study concludes that UA management led to higher forest carbon and net carbon balance (excluding carbon stock) compared to EA management in the eastern U.S. hardwood forests. Forest management strategies should be determined based on the ecological goal of increasing forest carbon stock and the economic goal of maximizing revenue from the timber market. The findings of this study have implications for policymakers and forest managers in mitigating climate change and carbon sequestration through sustainable forest management for timber production.

1. Introduction

Climate change caused by global warming is a major threat to ecosystems and human societies. Keeping climate change within safe limits would require net-zero greenhouse gas (GHG) emissions by the year 2050 (IPCC, 2018). The world's forests have absorbed 30 % (two petagrams per year) of annual anthropogenic CO_2 emissions (Bellassen and Luyssaert, 2014). Forest carbon is continuously cycled in carbon reservoirs and the atmosphere through biogeochemical processes and anthropogenic activities such as photosynthesis, respiration, decomposition of organic matter, deforestation, and wildfire. Harvesting does not immediately release biomass carbon into the atmosphere but transfers some carbon to the harvested wood products (HWPs) pool, where it remains stored for a long time. It was estimated that the HWPs pool in the U.S. forest sector had a net carbon uptake of 28 million metric tons in 2021 (EPA, 2023). In 2019, about 40 % of U.S. roundwood production was used in the manufacturing of durable HWPs, and two-thirds of that was used in the construction industry (Brandeis et al., 2021). Therefore, it is crucial to quantify the carbon in forests and forest products to evaluate the role of forests as carbon sinks and determine the forests' potential to mitigate climate change.

The forest carbon cycle comprises the biological cycle of forest ecosystems and the industrial cycle of forest products (Saud et al., 2013). The biological cycle represents the flow and exchange of carbon between the atmosphere, vegetation, and other organisms. During photosynthesis, trees absorb CO_2 from the air in a forest ecosystem and store the carbon as wood. During respiration, trees and other plants combine sugar and O_2 to produce energy for growth and release CO_2. The decomposition of organic matter leads to biogenic GHG emissions such as CO_2, CH_4, and N_2O (Johnson et al., 1995). The industrial cycle refers to the process by which forests are managed for commercial purposes. Deforestation caused by unsustainable and excessive harvesting practices, which involve felling trees at a rate that exceeds their natural regeneration, may diminish the ability of forests to absorb carbon, thereby increasing the concentration of CO_2 in the atmosphere. However, deforestation is not occurring in the United States. Even though the United States is the largest source of wood products in the world, its forests are continually increasing in wood volume (Oswalt et al., 2019; FAO–Food and Agriculture Organization of the United Nations, 2022). This shows that U.S. forest carbon stocks can be a sustainable source of renewable building materials that store carbon and avoid fossil carbon emissions (Brashaw and Bergman, 2021).

Forest carbon flux is the net change in carbon storage resulting from the balance between the carbon absorbed by forest growth and the carbon released due to natural and human disturbances. The forest management practices impact the ability of forests to sequester carbon, with variations influenced by stand heterogeneity. Intensive forest management such as utilizing fast-growing tree species for silviculture, mechanical land preparation, and fertilization enhance forest carbon sequestration capacity (Ameray et al., 2021). On the other hand, extensive management involving partial and selective harvesting can also promote carbon sequestration (Ameray et al., 2021). Sustainable forest management aims to balance current needs with preserving ecological, economic, and social values for future generations. The carbon flow from forests through HWPs is often considered carbon neutral if the carbon lost from harvests is recovered by tree regrowth on forest land over time through sustainable forest management practices (Lippke et al., 2011; Jakes et al., 2016; Brashaw and Bergman, 2021). This can result in a neutral net carbon flux from forests to industry.

Previous studies have modeled the forest carbon fluxes to assess the effects of forest management strategies on forest carbon balance. Eriksson et al. (2007) evaluated the impacts of forest management regimes, harvesting practices, and forest product use on carbon balances. How different management scenarios influence the overall carbon storage capacity of forest and wood products was explored by Liu and Han (2009). Ameray et al. (2021) reviewed the effects of three management regimes widely used in boreal, temperate, and tropical forests on carbon sequestration and stock dynamics. The results showed that specific forest management strategies could improve carbon sequestration. Forest management regimes, harvesting methods, and forest resource utilization all have impacts on net carbon emissions. Long-term carbon storage can be achieved through practices such as silvicultural, fertilization, and effective conversion of carbon into wood products.

The operations of harvesting machinery result in fuel and lubricant consumption, leading to embodied carbon emissions. Existing harvesting systems fall into categories of manual and mechanized, with specific harvesting decisions partly depending on the topography (Saud et al., 2013). Even with the relatively low GHG emissions and fossil energy demand of chainsaw-based manual systems, it's possible to balance the environmental impacts of increased mechanization due to improvements in fuel and emission control technologies (Berg and Karjalainen, 2003; Abbas and Handler, 2018). On the other hand, fuel consumption during different stages of harvesting can vary greatly. For example, skidding and yarding in redwood forests of northern California presented a higher environmental impact (Han et al., 2015), while logging, loading, and transport in intensively managed radiata pine harvesting contributed more to CO_2 emissions (Alzamora et al., 2022).

The eastern United States boasts an abundance of hardwood forests, which offers valuable ecosystem services and an extensive selection of wood products and value-added bioproducts. The diameter-influenced partial cuts and clear-cuts accounted for 80–90 % of timber removal in the east-central hardwood region (Luppold and Bumgardner, 2018). Divided by basal area removed, 55 % or less for uneven-aged (UA) management and 90 % or more for even-aged (EA) management (Luppold and Bumgardner, 2018). EA forest management involves stands with even tree ages, achieved through clearcutting and planting with a single species (Kuuluvainen et al., 2012). In contrast, UA forest management leads to stands with varying ages through selective cutting. UA management is more advantageous in preserving forest biodiversity and resilience but usually results in higher life-cycle environmental impacts on a production basis (Oneil et al., 2010; Lafond et al., 2014; Han et al., 2015; Oneil and CORRIM, 2021).

The production stage of wood products includes: extraction and upstream production (A1), transport to the factory (A2), and manufacturing (A3) (International Organization for Standardization (ISO), 2017). Previous hardwood lumber manufacturing (A3) research use hardwood forest resource (A1) life cycle inventory (LCI) data to develop cradle-to-gate wood product production including Puettmann et al. (2010), Hubbard et al. (2020), Alanya-Rosenbaum et al. (2021, 2022a, 2022b) and Heidari et al. (2023a, 2023b, 2023c). Therefore, developing detailed and comprehensive hardwood forest resource LCI data are needed for future life cycle assessment (LCA) and Environmental Product Declaration (EPD) development for hardwood products. This study aims to assess and analyze the carbon balance of hardwood forests and the environmental impacts, primarily Global Warming Potential (GWP), of harvesting systems in the eastern United States. The specific objectives include (1) simulation of current forest management scenarios using the forest vegetation simulator (FVS); (2) LCA of GHG emissions of manual and mechanized forest harvesting systems; and (3) carbon impact assessments of different phases by harvesting system. The resultant LCI data can be used to link hardwood log transport (A2) to hardwood lumber manufacturing (A3) to make data available to generate wood product EPDs (Bergman and Taylor, 2011; Ibáñez-Forés et al., 2016; International Organization for Standardization (ISO), 2017; ULE, 2019).

2. Data and methods

2.1. Study area

Four subregions were defined based on the various stand types and species composition as northeast (NE: Connecticut, Massachusetts, Maine, New Hampshire, Rhode Island, and Vermont), mid-Atlantic (MA: Kentucky, North Carolina, South Carolina, Tennessee, Virginia, Delaware, Maryland, New Jersey, New York, Ohio, Pennsylvania, and West Virginia), southeast (SE: Alabama, Florida, Georgia, Mississippi, and Arkansas), and northcentral (NC: Illinois, Indiana, Iowa, Michigan, Minnesota, Missouri, and Wisconsin). The different stand types in the four regions result from an interplay between climate, soil, topography, and historical factors (Schoennagel et al., 2004).

2.2. Data

The regional base dataset was compiled using the U.S. Forest Service's Forest Inventory and Analysis National Program (FIA). To generate region-specific reports, the Design and Analysis Toolkit for Inventory and Monitoring (DATIM), a tool within the FIA, was employed. This process involved applying DATIM individually to each region, selecting all the states within that region for inclusion in the report. The creation of the base dataset relied on two reports obtained through this approach. The first report provided data on the net volume of live trees, measuring at least five inches in diameter at breast height or diameter at root collar, expressed in cubic feet. This data was categorized by forest type group and stand-size class. The second report contributed information on the average annual harvest removals of sawlog volume from sawtimber trees, measured in board feet according to the international ¼-inch rule. This data was categorized by owner and species group. The international ¼-inch rule is a standard derived from an analysis of losses during the conversion of sawlogs to lumber. This rule accounts for taper, assuming a taper of ½ inch in four feet. All calculated values under this rule are rounded to the nearest five board feet.

The average annual harvest removals included species, species groups, and landowner classifications, encompassing federal, state, county, local governments, and private land ownership. Softwood species were excluded from summary statistics. The data was incorporated into an Excel database and converted from board feet to cubic meters using a conversion factor of 0.0045 (Spelter, 2004). Cubic feet of trees in each species or species group were determined based on net live tree volume and stand size class, as defined by the FIA user guide (Burrill et al., 2021). Softwood species were excluded from summary volume calculations. These calculations were carried out individually for each region and integrated into the Excel spreadsheet for reference and additional analysis. FIA data, essential for FVS utilization, were accessed through the FIA data management system, covering all states within the four regions.

Adhering to the ISO 14044 (International Organization for Standardization, 2006) guidelines, data on harvesting machine productivity and fuel (diesel and lubricants) consumption were collected from peer-reviewed publications and technical reports to be integrated into the LCI. The selection criteria prioritized recent data (within five years), while data older than ten years were excluded. Geographical alignment with the four defined subregions in this study was a crucial criterion to ensure the relevance and applicability of the data for assessing GHG emissions. The collected raw data were categorized into two primary groups: manual and mechanized harvesting systems, as detailed in Tables S2-S5. Subsequently, the average productivity and fuel consumption for both manual and mechanized systems within the four regions were input into the SimaPro software to calculate CO_2e emissions. As a result, the differences in fuel consumption among regions can be attributed to the spatial attributes of the harvesting plots in the subregions.

2.3. Methods

2.3.1. Forest vegetation simulator

FVS is a widely used forest growth model in the United States, operating at the stand level but with broad spatial and temporal capabilities. It begins with inventory data and adjusts predictions using self-calibration. FVS models individual tree growth, disturbances like pests and fire, and allows for management rule specification (Crookston and Dixon, 2005). Its applications range from single stand prescriptions to landscape assessments, addressing issues such as forest development, wildfire risk, and pest management. However, FVS primarily focuses on the ecological aspects of forests and does not inherently consider external factors or simulate broader land use changes driven by agricultural demand or other non-forest land use changes.

The simulation spanned from 2020 to 2040, with the starting year chosen to align with the availability of the FIA data. The state-level runs were conducted in accordance with the FVS variants, tailored specifically to the NE, MA, SE, and NC regions, accounting for geographic differences among these areas (Hoover and Rebain, 2011). Two scenarios were created: a UA management utilizing thinning or partial harvest, and an EA management scenario utilizing clearcutting. EA and UA management were applied to identical sets of plots in each region, with the NE, MA, SE, and NC regions having a total of 1226, 6692, 4637, and 9596 plots, respectively. For the simulation of harvest, 200 plots were selected in each state, and these same plots underwent simulated harvesting under both management scenarios.

In the United States, the clear-cutting rate in forest harvesting was approximately 40 %, while 60 % of the total harvested area was achieved through partial cutting methods (Masek et al., 2011). However, the percentages of clear-cutting and partial harvesting may vary by region. In the SE region, out of 1.18×10^{10} m^2 of forested land, 35 % were harvested through clear-cutting, while the remaining 65 % underwent partial cuts (Siry and Cubbage, 2005). Similar data have been reported for states in the MA region, including North Carolina, South Carolina, and Virginia. In a total forested area of 3.46×10^9 m^2 across these states, 84 % were harvested through clear-cutting, while the remaining 16 % underwent partial cuts (Siry and Cubbage, 2005). In West Virginia, the most commonly used harvesting methods were thinning/diameter limit cutting (62 %) and partial harvesting/shelterwood/seed tree (27 %), with clear-cutting used less frequently (9 %) (Milauskas and Wang, 2006). Belair and Ducey (2018) reported the percentages of harvesting methods in six states in the NE region. Clear-cutting, including commercial clearcuts/seed tree/clearcut with reserves/silvicultural clearcut/group/patch selection, accounted for 28 %. Thinning, which includes crown thinning/crop tree release/low thinning/shelterwood prep cut/overstory removal/single tree/group selection/pre-commercial/geometric thinning, also accounted for 28 %. The remaining categories, incidental harvest, shelterwood, and high grade, made up 19 %, 14 %, and 10 %, respectively. Another study covering four states in the NC region revealed that 64 % of the plots were harvested with partial cuts, while 9 % were harvested with clearcuts. Thinning, small diameter, and other cuts accounted for 5 %, 14 %, and 8 %, respectively (Luppold and Bumgardner, 2018).

Rotation ages also differ based on the region and the main species groups being managed. According to the silvicultural guide for northern hardwoods in the NE, a rotation age of 107 to 119 years, which is a reasonable minimum rotation age for quality sawtimber products, is recommended (Leak et al., 2014). Considerations for other species groups include rotation ages of 80 to 100 years for paper birch, maple, and beech, and an even shorter rotation age of 40 to 60 years for aspen-birch species groups (Leak et al., 2014). In the case of southern hardwoods, a recommended rotation age of 80 to 100 years is advised (Rousseau, 2019). In the central hardwoods region, rotation ages range from 70 to 120 years for hardwood species (Wisconsin Department of Natural Resources, 2020).

Within FVS, specific components can be incorporated into simulations, including regeneration methods for UA and EA. For the UA method, the component can thin a stand based on a Q-factor. The Q-factor, when multiplied by the desired number of trees in a diameter class, provides the number of trees in the adjacent class of smaller diameter trees (Dixon, 2002). The determination of the Q-factor is guided by the condition of "stocking above the percentage of normal," with the normal stocking level set at 65 %. For EA, the component can apply a clearcut/coppice based on the same condition, triggering a harvest when met. Additional UA and EA settings can be found in the Table S1. Using the "stocking above 65%" condition serves as a control to facilitate direct method comparison, avoiding the need for annual harvest scheduling, which may lead to harvesting understocked stands. This approach aligns with real-world harvesting practices that activate when a stand achieves a desired stocking level, ensuring practicality and economic feasibility for managers. The choice of 65 % as the normal stocking level was based on hardwood stocking charts, where percentages of 60–100 are considered fully stocked (Bennett and Good Forestry of the Granite State Steering Committee, 2010). Stands around 100 % stocking are considered overstocked for timber management, with slow tree growth and high natural mortality, primarily among small trees (Clark and Hutchinson, 1989).

Within FVS, state-level simulations simultaneously generate carbon estimates, facilitated by "The Fire and Fuels Extension." This extension models fuel dynamics and potential fire behavior over time within the context of stand development and management (Rebain et al., 2010). The carbon calculations within this extension involve converting carbon content in living and dead biomass to carbon units by applying a factor of 0.5, as described by Penman et al. (2003). Similarly, litter and duff biomass were converted using a multiplier of 0.37, in accordance with Smith and Heath (2002). As Hoover and Rebain (2011) noted, the Fire and Fuels Extension module of FVS did not account for soil carbon, which includes all roots of dead trees and fine roots of live trees.

2.3.2. *Life cycle assessment*

The LCA system boundary is illustrated in Fig. 1, encompassing the initial four harvesting phases at site: felling, extraction, processing (topping and delimbing), and loading. The FVS component provides data to the LCA component, including forest carbon stock, carbon growth, and carbon removal. To ensure analytical consistency, the functional unit was standardized to 1 m^3 of hardwood harvested, extracted and processed at the landing. SimaPro 9, utilizing industry-specific databases, was employed for conducting the LCA. The input parameters included the quantity of diesel (in liters) and proxy lubricant (in kilograms) consumed for harvesting 1 m^3 of hardwood. SimaPro utilized the 100-year time horizon GWP to convert GHG emissions into equivalent CO_2 emissions.

3. Results and discussion

3.1. *Forest carbon stock*

The Stand Carbon Report covers the major carbon pools defined by the U.S. Carbon Accounting Rules and Guidelines and the IPCC Good Practice Guidance. These include aboveground live trees, belowground live trees (coarse roots), belowground dead trees, standing dead trees, down dead wood, forest floor, and understory (shrubs/herbs) (Hoover and Rebain, 2011). Above ground carbon includes the total above ground live trees, including stems, branches, and foliage, but excluding roots, herbs, and shrubs. Below ground carbon includes the belowground live roots of live trees and the belowground dead roots of dead and cut trees. Deadwood carbon consists of standing dead trees, including stems and any branches and foliage still present, but excluding roots, and forest down dead wood which includes all woody surface fuel regardless of size (Rebain et al., 2010). The carbon stock values at five-year intervals starting from 2020 were averaged for both the EA and UA management scenarios. The UA management scenario had larger averages for all regions than the EA scenario. In the UA management

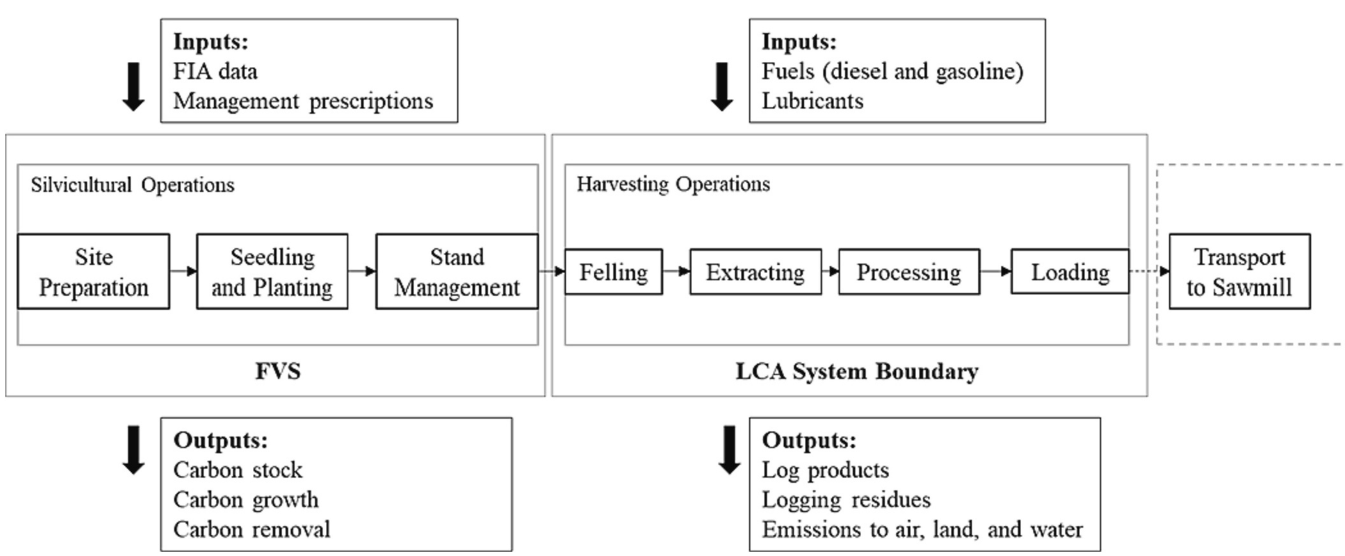

Fig. 1. Integrated FVS and LCA System boundary and process flow diagram.

scenario, the NE, MA, SE, and NC regions had averaged 8.14, 6.04, 6.34, and 3.14 kg/m^2 of carbon, respectively. In the EA management scenario, the NE, MA, SE, and NC regions had averaged 7.07, 5.11, 6.05, and 2.76 kg/m^2 of carbon, respectively. The initial carbon stocks in each region vary between the two management scenarios due to the application of distinct prescription practices in 2020, with partial harvesting in UA management and clear-cutting in EA management. A side-to-side comparison can be seen in Table 1. A paired *t*-test with a sample size of four was performed to determine whether significant differences existed in the average carbon stocks between UA and EA management scenarios across four regions. The calculated *p*-value was 0.04 at a 5 % significance level, leading to the rejection of the null hypothesis, indicating a significant difference in carbon stocks between UA and EA management in at least one of the regions. From the single standpoint of increasing forest carbon, this result indicates that UA management is the better choice. Due to the nature of EA management, this result is not surprising, given the 20-year timeline. Harvests under EA management remove almost all the trees within a very short period (Leak et al., 2014). Therefore, it would take a longer amount of time to regrow the carbon stock under this management scenario compared to the UA scenario, where more residual trees are left per unit area. The UA management removes trees in various size classes to maintain a sustained yield (Leak et al., 2014). This translates to a sustained residual carbon stock that would have otherwise been removed in the EA scenario. However, the difference between these two management scenarios may be negated given a longer timeline where the EA stand would be able to regenerate more carbon.

The location of carbon within the stands varied slightly between the two management scenarios. In both scenarios, the largest percentage of carbon was located in the "above ground merchantable trees" category. The UA management scenario had the largest percentages of carbon stored in above ground merchantable trees, forest floor, below ground live and down dead wood, with 42 %, 16 %, 16 %, and 14 %, respectively. The remaining carbon was stored in below ground dead, standing dead, and shrub/herb. The EA management scenario had the largest percentages stored in above ground merchantable trees, down dead wood, forest floor, and below ground live, with 31 %, 20 %, 18 %, and 15 %, respectively. The remainder was stored in below ground dead, standing dead, and shrub/herb. The average kilograms of carbon per square meter were calculated for four different locations based on the FVS outputs. The four locations were above ground, below ground, deadwood, and litter. Fig. 2 is organized to provide a side-by-side comparison of average carbon stocks between the two management scenarios.

Table 1
Carbon stock, growth, and removals of hardwood forests in the eastern U.S.

	Northeast	Mid Atlantic	Southeast	North Central	Average All Regions
Carbon Stock All Years Average (kg/m^2)					
Uneven Aged Scenario	8.14	6.04	6.34	3.14	5.91
Even Aged Scenario	7.07	5.11	6.05	2.76	5.25
Total Carbon Growth (kg/m^2)					
Uneven Aged Scenario	2.38	1.90	2.96	0.79	2.01
Even Aged Scenario	1.98	1.48	2.80	0.74	1.75
Average Carbon Growth Between Cycles (5 years) (kg/m^2)					
Uneven Aged Scenario	0.59	0.48	0.74	0.20	0.50
Even Aged Scenario	0.49	0.37	0.70	0.18	0.44
Carbon Ratio: Above-ground/Below-ground					
Uneven Aged Scenario	8.50	7.51	7.87	5.78	7.41
Even Aged Scenario	5.72	6.01	6.21	6.19	6.03
Total Carbon Removals (kg/m^2)					
Uneven Aged Scenario	0.20	0.31	0.51	0.15	0.29
Even Aged Scenario	0.91	0.84	1.13	0.25	0.78

3.2. Forest carbon growth

Forest carbon growth was calculated for both the total growth from the first year to the last year of the simulation (2020–2040) and the amount of growth in five-year increments. The carbon growth from 2020 to 2040, as well as the carbon growth in five-year increments, was larger for all regions in the UA scenario than the EA scenario. A paired *t*-test, conducted with a sample size of four, aimed to assess if significant differences were present in the average total carbon growth between UA and EA management across four regions. With a calculated *p*-value of 0.06 at a 5 % significance level, the null hypothesis was not rejected, indicating no significant differences in total carbon growth between UA and EA management. In a second paired *t*-test with a sample size of four, average carbon growth at five-year intervals between UA and EA scenarios across four regions was assessed. The null hypothesis was retained, as the calculated *p*-value of 0.06 at a 5 % significance level indicated no significant differences in five-year carbon growth between the two scenarios. The detailed values for each region and scenario can be found in Table 2.

The percent change in each classification of carbon was calculated for both regions from the starting year of 2020 to the ending year of 2040. In both scenarios, the three largest categories of carbon growth were the sum of aboveground total live, the sum of belowground live, and aboveground merch live. The EA scenario showed percentage increases of 113 %, 110 %, and 70 % respective to category, while the UA scenario had increases of 86 %, 84 %, and 69 %, respectively. The EA scenario exhibited larger percent increases in these three categories than the UA scenario.

3.3. Forest carbon removal

Comparison between the UA and EA methods resulted in a relatively similar amount of total harvest volume (Table 3). The UA management scenario resulted in 1.27 × 10^8 m^3, while the EA method resulted in 1.35 × 10^8 m^3 for a difference of 8 × 10^6 m^3. In the EA management scenario, there were 8.53 × 10^7 m^3 of sawlog (63 % of the total) and 5.02 × 10^7 m^3 of pulp (37 %). In the UA scenario, there were 8.41 × 10^7 m^3 of sawlog (66 % of the total) and 4.28 × 10^7 m^3 of pulp (34 %). The NE and SE both had higher harvest totals under the UA management scenario, with 1.19 × 10^6 and 2.82 × 10^6 m^3 higher respectively. On the other hand, the MA and NC regions both had higher harvest totals under the EA management scenario, with 9.97 × 10^6 and 2.62 × 10^6 m^3 higher respectively. The sawlog harvest followed a similar trend, with the NE and SE having larger harvest totals under UA management, while the NC and MA had higher harvest totals under EA management. For pulpwood harvests, all regions except the NE had higher harvest totals under EA management.

The EA management scenario had larger kilograms of carbon per square meter removals for each reporting year in the simulation than the UA management scenario. The EA management scenario showed a decreasing trend in carbon removals, with the highest removals occuring in 2020 and the lowest in 2040. In contrast, the UA management scenario showed a peak of removals in 2025, followed by a steady decrease until 2040. In a paired *t*-test, significant differences were found in annual average carbon removals between UA and EA management, with

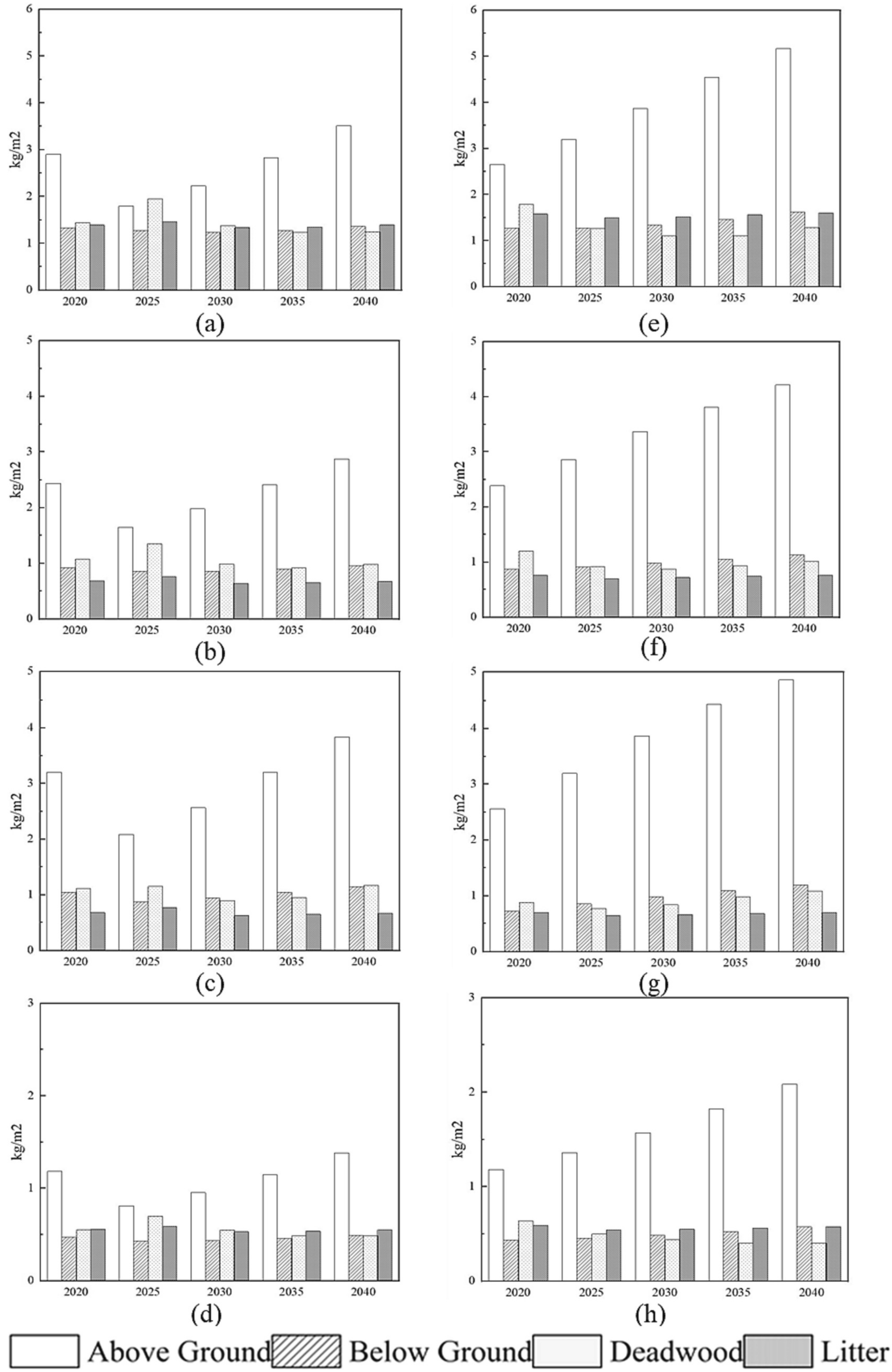

Fig. 2. Carbon stocks for different regions and management scenarios. (a): NE region, even-aged management; (b): MA region, even-aged management; (c): SE region, even-aged management; (d): NC region, even-aged management; (e): NE region, uneven-aged management; (f): MA region, uneven-aged management; (g): SE region, uneven-aged management; (h): NC region, uneven-aged management.

Table 2
Carbon stock for all regions and management scenarios.

	Northeast	Mid Atlantic	Southeast	North Central	Average All Regions
Regional Average All Years (kg/m^2)					
Uneven Aged Scenario	8.14	6.04	6.34	3.14	5.91
Even Aged Scenario	7.07	5.11	6.05	2.76	5.25
Regional Average 2020 (kg/m^2)					
Uneven Aged Scenario	7.30	5.22	4.87	2.84	5.06
Even Aged Scenario	6.49	4.62	4.88	2.53	4.63
Regional Average 2040 (kg/m^2)					
Uneven Aged Scenario	9.67	7.13	7.83	3.64	7.07
Even Aged Scenario	8.46	6.11	7.68	3.26	6.38
Percentage Increase 2020–2040					
Uneven Aged Scenario	33 %	36 %	61 %	28 %	38 %
Even Aged Scenario	30 %	32 %	57 %	29 %	36 %

Table 3
Carbon removals for both management scenarios.

	2020	2025	2030	2035	2040	Regional Average
Uneven Aged Carbon Removals (kg/m^2)						
Northeast	0.07	0.05	0.02	0.03	0.04	0.04
Mid Atlantic	0.08	0.09	0.05	0.05	0.03	0.06
Southeast	0.04	0.20	0.13	0.07	0.08	0.10
Northcentral	0.01	0.04	0.04	0.03	0.03	0.03
Yearly Average	0.05	0.09	0.06	0.05	0.04	0.06
Even Aged Carbon Removals (kg/m^2)						
Northeast	0.61	0.11	0.07	0.04	0.07	0.18
Mid Atlantic	0.44	0.14	0.12	0.08	0.06	0.17
Southeast	0.40	0.27	0.22	0.13	0.10	0.23
Northcentral	0.01	0.07	0.07	0.06	0.04	0.05
Yearly Average	0.34	0.14	0.12	0.08	0.06	0.15

a p-value of 0.04 at 5 % significance level (rejected the null hypothesis). Furthermore, in a second paired t-test, the combined harvest totals (pulp and sawlog) for UA and EA management scenarios were analyzed across four regions. The null hypothesis was upheld, with a calculated p-value of 0.45 at a 5 % significance level, suggesting no significant differences in averaged harvest totals between the two scenarios. Despite the EA management scenario having higher total harvest and carbon removal than the UA management scenario, statistical tests revealed significant differences only existed in carbon removal between two management scenarios, with no significant distinctions found in total harvest across various scenarios. This suggests that the UA management scenario would be the better choice from the standpoint of carbon and harvest totals since it results in a statistically significant lower amount of carbon being removed and a statistically insignificant lower amount of total board feet being harvested.

3.4. Fuel and lubricant consumption

Manual and mechanized ground-based systems are the two most used harvesting systems in the eastern United States. The manual ground-based system involves logging with chainsaws and extracting with cable skidders. The process typically begins with selecting mature trees to be felled, delimbing and bucking after the trees are felled, and then collecting the logs using cable skidders. While manual harvesting can be labor-intensive and time-consuming, it is often applied in areas where machinery is unavailable. Nonetheless, manual harvesting tends to be a less disruptive method as it allows for more selective harvesting with less disturbance and damage to the forest. The mechanized ground-based system involves the application of heavy machinery, including feller-bunchers and grapple skidders, harvesters and forwarders, to carry out the various harvesting phases. Specialized machines such as harvesters are designed to efficiently cut, debark, and process the timber to the desired length in a single pass. After the logs have been transferred to a central location, they undergo sorting by species, size, and grade before being processed into multiple forest products, including lumber and pulp. Despite the greater efficiency compared to manual harvesting, mechanized harvesting may cause negative environmental impacts, such as soil compaction and damage to adjacent trees or vegetation. However, the negative effects can be mitigated through appropriate planning and management to promote sustainable commercial harvesting. Machine productivity, fuel, and lubricant consumption of manual and mechanized harvesting systems were calculated across four regions and under different management practices (Table S2-S5).

It was found that the same configuration mechanized system had higher productivity and lower fuel consumption rate for hardwood clearcutting than partial harvesting in the NE region. This indicates that mechanized systems are more fuel-efficient for clearcutting, especially during felling and extraction. Conversely, manual systems consumed more fuel when partial harvesting was performed than mechanized systems on average, with 4.27 and 2.39 L/m^3 of diesel and 0.08 and 0.03 L/m^3 of lubricant, respectively. Similar results were observed in the MA region, where mechanized systems were also more productive for clearcutting than partial harvesting. However, the fuel consumption rate did not follow the same trend, with mechanized systems exhibiting higher fuel consumption for clearcutting. Fewer differences were found in the average total fuel consumption for the manual systems for different harvest types. Nonetheless, the average total diesel consumption was higher for the mechanized system than for the manual system when clearcut was conducted, with 4.58 and 3.35 L/m^3, respectively. In the NC region, fuel efficiency and productivity were higher for manual systems during partial harvesting. Conversely, mechanized systems were more fuel-efficient for clearcut. Fig. 3 depicts the total fuel consumption averaged by different combinations of impact variables with interquartile range, range with 1.5IQR, median line, and outliers. It is evident that the mechanized system outperformed the manual system in terms of harvesting productivity. Additionally, the productivity of mechanized systems for partial harvesting was lower than that of clearcut. Han et al. (2015) also indicated that hourly harvesting production rates for thinning operations were lower than for clearcutting operations. It was also observed that UA management relying on partial harvesting led to higher fuel and lubricant consumption compared to evenly managed clearcut scenarios, which aligns with the previous research findings (Han et al., 2015). Overall, the average total fuel consumption for manual hardwood harvesting was estimated at 3.62 L/m^3, with a total lubricant consumption of 0.22 L/m^3. By contrast, the mechanized harvesting system exhibited lower energy consumption, with average total fuel consumption estimated at 3.00 L/m^3 and total lubricant consumption at 0.14 L/m^3. The study conducted by Oyier and Visser (2016) also reported an average fuel consumption of 3.18 L/m^3 for the ground-based timber harvesting system.

3.5. GHG emissions

The relatively high GHG emissions were observed in the NE when the manual system performed partial harvesting. This distinction from the results of previous studies indicates that actual GHG emissions depend on a variety of factors, such as equipment type, extraction distance, and operator skills. Manual systems also produce substantial GHG emissions, especially when using cable skidders. The reduced environmental

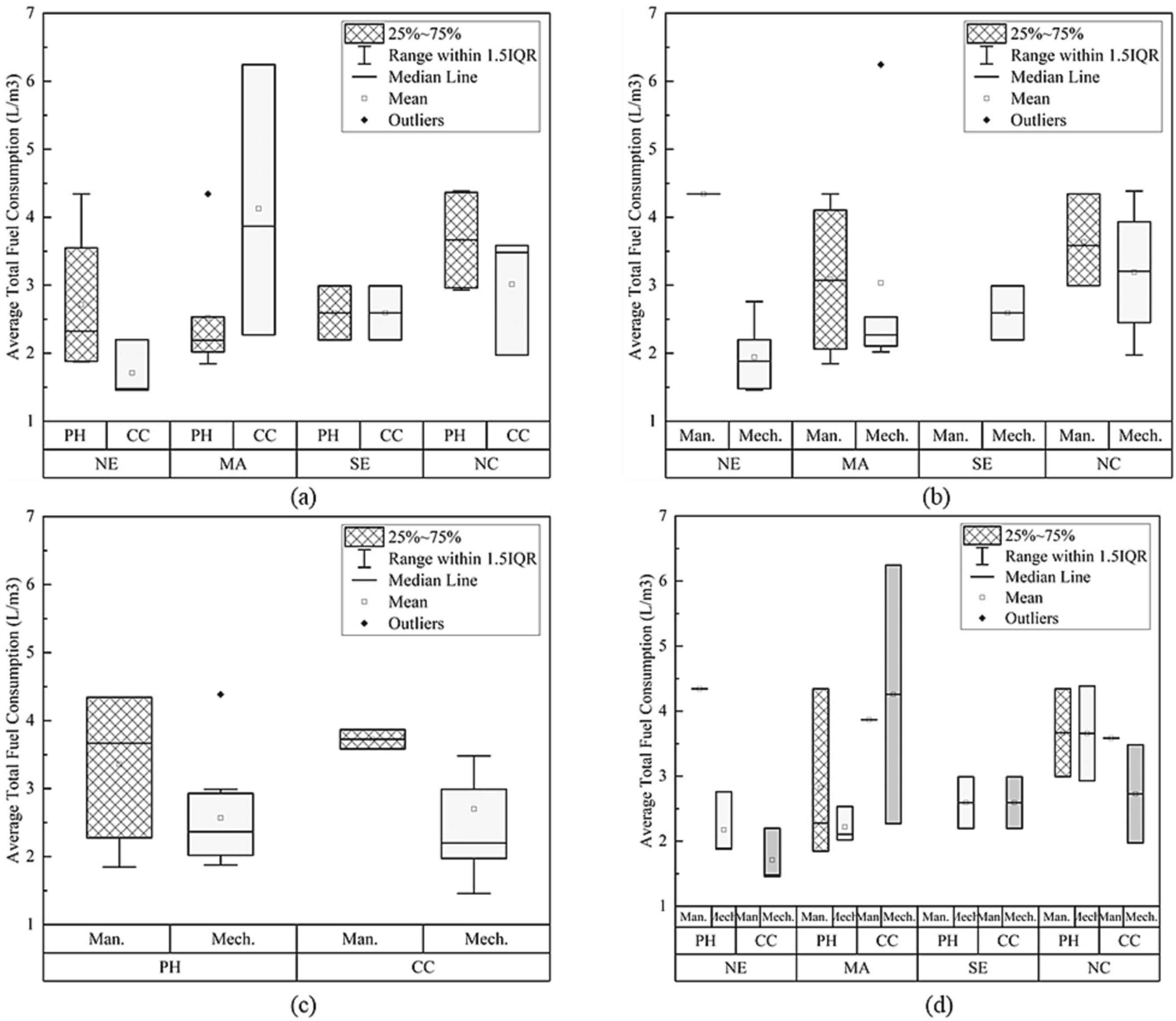

Fig. 3. System, harvest type, and spatial differences in average total fuel consumption. (a): total fuel consumption averaged by region and harvest type; (b): total fuel consumption averaged by region and system; (c): total fuel consumption averaged by harvest type and system; (d): total fuel consumption averaged by region, harvest type, and system. PH: partial harvest, CC: clearcut. Man.: manual, Mech.: mechanized.

impact of mechanized systems is due to a 55.63 % decrease in fuel consumption for the extraction process compared to the manual system. In the MA region, GHG emissions from partial harvesting by manual systems were slightly greater than clearcut. While the low level of GHG emissions of mechanized systems from partial harvesting is attributed to the missing loading data. The GHG emissions from partial harvest and clearcut in the NC manual systems were also similar. The main reason for the differences in the NC mechanized systems are still the fuel consumption gap during extraction. In the SE region, data limitations prevented a breakdown of environmental impacts under specific management. Table 4 shows the detailed GHG emissions for different harvesting phases.

Overall, the results indicate that within the mechanized systems, the fuel consumption for felling and extraction is lower for clearcut than for partial harvest, mainly because partial harvesting increases the distance travelled by the machinery to the target. Regional variations in GHG emissions may be associated with different harvesting systems, but it is uncertain whether manual or mechanized systems have higher emissions. When fuel-saving equipment is adopted or the extraction distance is shorter, less energy will be consumed for manual harvesting. In contrast, mechanized harvesting can use specialized machinery such as feller-bunchers, skidders, and processors powered by diesel engines to operate more efficiently. Many variables can affect fuel consumption rates in both manual and mechanized harvesting systems, including the scale and complexity of the operation, the terrain, the tree species being harvested, and the skill level of the harvesting crew. Regarding the harvesting process, the extraction and loading processes contribute more to GHG emissions, followed by felling and processing. The comparison of GHG emissions between manual and mechanized systems at different harvesting stages has shown in Fig. 4. According to the results, the contribution of GHG emissions from manual systems under UA management for extraction and loading was 59.5 % and 34.5 %, respectively. However, felling and processing processes only contributed to 5.4 % and 0.6 %. Similarly, 49 %, 38 %, and 13 % of GHG emissions came from extraction, loading, and felling from manual systems under EA management. The mechanized system, on the other hand, had a higher percentage of GHG emissions during the felling stage under both management modes. In particular, it contributed 36.05 % and

Table 4
GHG emissions from manual and mechanized harvesting systems under uneven-aged and even-aged management scenarios (kg CO_2e/m^3).

		Manual ground-based system				Mechanized ground-based system			
		Partial harvest		Clearcut		Partial harvest		Clearcut	
		Fuel	Lube	Fuel	Lube	Fuel	Lube	Fuel	Lube
Northeast	Felling	0.232	0.001			1.540	0.014	0.671	0.014
	Extraction	8.680	0.047			3.850		2.320	
	Processing	0.073				1.400	0.013	1.580	0.013
	Loading	4.560	0.025			0.798		0.785	
	Total	13.62				7.62		5.38	
Mid-Atlantic	Felling	0.479	0.003	0.603	0.004	2.070	0.272	3.870	0.211
	Extraction	5.700	0.362	5.460	0.470	2.940	0.250	5.450	0.262
	Processing	0.073				0.931		0.656	0.002
	Loading	4.560	0.025	4.560	0.025			4.560	0.025
	Total	11.20		11.12		6.46		15.04	
Southeast	Felling					4.27		4.27	
	Extraction					2.58		2.58	
	Processing					1.56		1.56	
	Loading					1.20		1.20	
	Total					9.61		9.61	
Northcentral	Felling	1.230	0.001	2.340		5.990		3.860	
	Extraction	6.760	0.047	5.110		4.410		2.830	
	Processing	0.073							
	Loading	3.500	0.025	3.930		2.440		3.930	
	Total	11.64		11.38		12.84		10.62	

30.82 % of GHG emissions, respectively. Nevertheless, the extraction process remained responsible for the higher proportion of GHG emissions, with 35.74 % and 32.06 % for each mode. The processing and loading processes accounted for 13.24 % and 14.97 % of GHG emissions during partial harvesting by mechanized systems. By contrast, during clearcutting, these processes contributed 12.31 % and 24.81 %, respectively. However, it's worth noting that mechanized systems may have higher initial capital costs and require more maintenance and repairs than manual systems, so a cost-benefit analysis should be conducted to determine the most appropriate harvesting system for a particular operation.

3.6. Forest carbon balance

During the simulation period from 2020 to 2040, it was found that the average carbon stock in eastern U.S. hardwood forests under UA and EA management was 5.91 and 5.25 kg/m^2, respectively. This result suggests that UA strategies lead to longer rotation periods and increase carbon accumulation time in the forest ecosystem. The average carbon stock in the four subregions followed the same trend, but there were spatial variations depending on factors such as climate conditions and forest type. For instance, the average carbon stock in NE, MA, SE, and NC regions under the UA scenario was 8.14, 6.04, 6.34, and 3.14 kg/m^2, respectively. In contrast, the average carbon stock estimated when EA practices were applied was 7.07, 5.11, 6.05, and 2.76 kg/m^2, respectively. The difference in rotation periods between the two management regimes also contributes to the prediction variance in average carbon growth. Results of the study indicated that average carbon growth under UA management was 2.01 kg/m^2 in the eastern United States and 2.38, 1.90, 2.96, and 0.79 kg/m^2 in the subregions of NE, MA, SE, and NC, respectively. However, the average carbon growth for all areas of the EA management scenario was 1.75 kg/m^2, with subareas averaging 1.98, 1.48, 2.80, and 0.74 kg/m^2. The impact of other factors, such as climate and harvest intensity, can also vary among regions, potentially influencing the differences in carbon growth rates.

It's clear that harvesting methods have a significant impact on carbon removal in eastern U.S. hardwood forests. Specifically, clearcut produced higher carbon removals than partial harvesting, with an average of 0.78 kg/m^2 compared to 0.29 kg/m^2. However, it's noteworthy that there were differences in carbon removal rates among regions. Under UA management, the average carbon removal in the subregions was 0.20, 0.31, 0.51, and 0.15 kg/m^2. Whereas under EA management, these rates were 0.91, 0.84, 1.13, and 0.25 kg/m^2, respectively. One factor that could contribute to this variability is the natural diversity of climate and forest ecosystems across regions. In particular, forests in regions with higher precipitation and soil organic matter content may have greater carbon removal potential due to faster growth rates and higher biomass accumulation. Other factors that may influence carbon removal potential include forest management practices and policy and market drivers, such as promoting the use of low-carbon intensity products or renewable energy.

According to the FIA statistics, the eastern U.S. hardwood forests were estimated to cover approximately 8.79×10^{10}, 4.50×10^{11}, 1.84×10^{11}, and 2.76×10^{11} m^2 in the NE, MA, SE, and NC regions. The forest carbon balance was defined as the sum of carbon stocks and carbon growth, subtracted by carbon removal and carbon emissions. In contrast, the net carbon balance was calculated by subtracting carbon removal and carbon emissions from carbon growth. The forest carbon balance of eastern U.S. hardwood forests under UA management regimes was estimated to be 7.73 kg/m^2, with a net carbon balance of 1.69 kg/m^2. In comparison, the carbon balance under EA management was estimated to be 6.20 kg/m^2, with a net carbon balance of 0.91 kg/m^2. The carbon balance between carbon emissions and carbon release at the time of harvesting was also calculated. According to the analysis results, this carbon balance was positive for hardwood forests in the eastern US, indicating that total carbon release exceeded total carbon emissions under both UA and EA management, approximately 3×10^{11} and 7.35×10^{11} kg, respectively.

4. Conclusions

The NE and SE regions both had almost entirely increased harvest, biomass, and carbon totals under UA management except for pulpwood in the SE and below ground carbon in the NE. Thus, these regions are better suited for UA management. On the other hand, the MA and NC regions both had lower harvest totals but higher biomass and carbon averages under the EA management scenario. These outcomes suggest that the choice of management method may depend on objectives and goals. In cases where harvest productivity is prioritized, EA management demonstrates better results, whereas UA management may be more suitable when biomass and carbon is a priority. For all regions the UA management scenario showed increased carbon and biomass compared

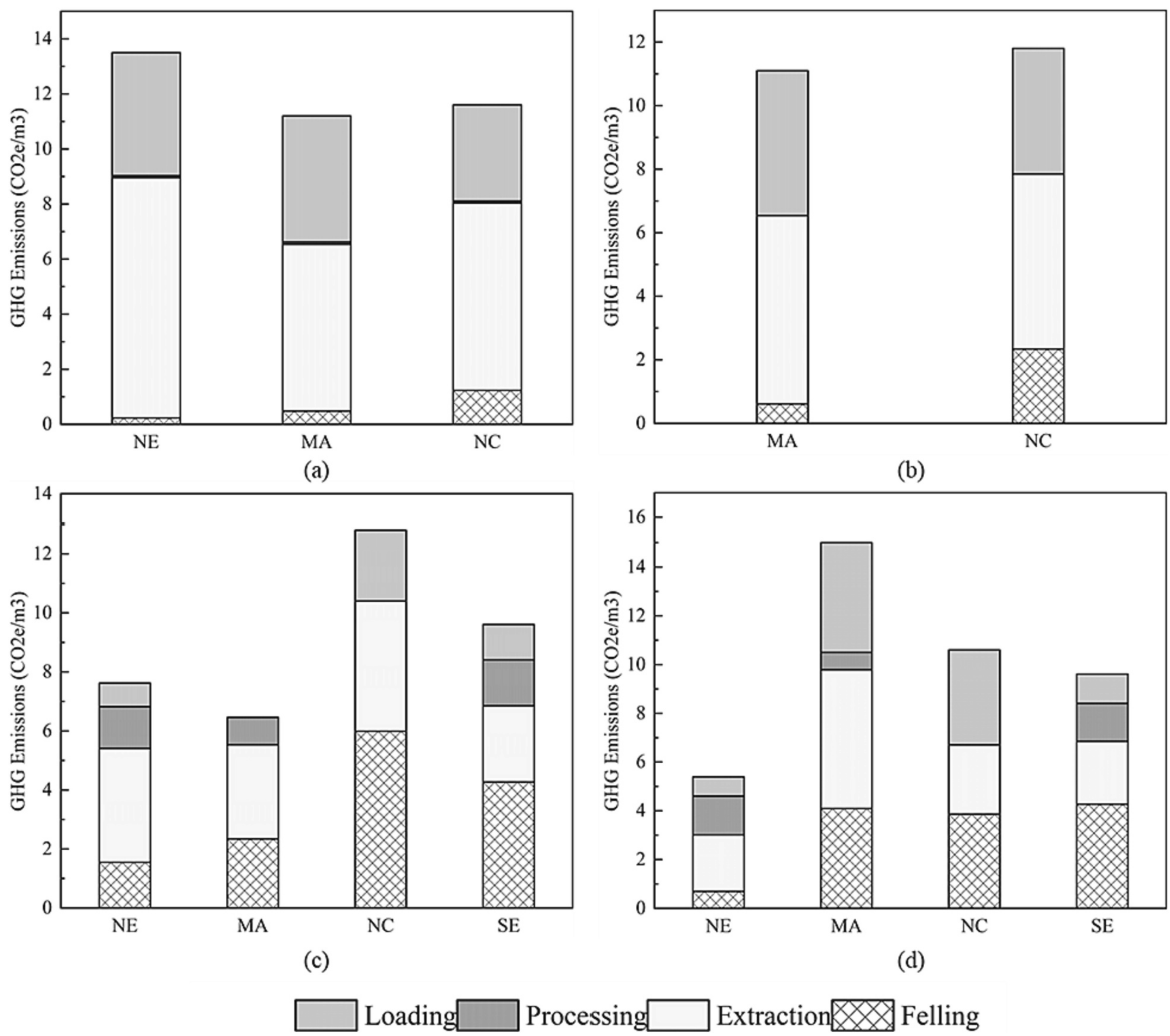

Fig. 4. Comparison of GHG emissions between manual and mechanized systems at various harvesting stages in subregions. (a): manual-PH; (b): manual-CC; (c): mechanized-PH; (d): mechanized-CC. PH: partial harvest, CC: clearcut.

to EA indicating that from a carbon sequestration standpoint UA management may be the preferred choice. In the eastern United States, the availability of hardwood forests for carbon sequestration amounts to 9.99×10^{11} m^2. However, the actual acreage and net timber volume of hardwood forests are contingent upon multiple factors, such as stand age structure, health, soil and climate type, and forest management practices. Both partial harvest-led UA management and clearcut-led EA management can affect the carbon sink in hardwood forests. Compared to clearcutting, low-intensity or partial harvesting techniques offer the potential to sustain forest carbon stocks while minimizing the likelihood of disturbance, enabling the utilization of the forest for wood products or biomass energy production (Harmon et al., 2009; McKinley et al., 2011). Despite harvesting-related carbon emissions, hardwood forests have not experienced significant losses in carbon stocks and carbon balances. Regarding productivity and fuel consumption, while differences exist between manual and mechanized harvesting systems, the average GHG emissions range from 9 to 12 kg CO_2e/m^3 of hardwood timber and are not significantly different between the two systems. These results are comparable to other U.S. forest resource LCAs including Han et al. (2015) and Oneil and Puettmann (2017). Forest resources consume 63–187 MJ of energy to produce 1 m^3 of hardwood sawlogs. It is important to note that emissions from fuel and lubricant combustion during harvesting are substantially lower than the carbon stored in hardwood timber and forest logging residues. These LCA outcomes can be used as upstream processes for hardwood manufacturers interested in developing EPDs for products that use these resources as inputs such as lumber, pallets, and flooring.

Policymakers can utilize these findings by employing a multi-objective optimization model that considers forest growth in specific regions and annual wood demand, with data sourced from the FVS, local timber markets, and the wood processing industry. They can define objective functions to balance meeting wood demand and ecological goals, including carbon stock increase, carbon sequestration, biodiversity, and ecosystem services. By assigning weights to these objectives and setting constraints aligned with ecological considerations, policymakers can use the model to find solutions that strike a balance between economic and ecological objectives. Furthermore, quantitative methods, such as estimating timber market revenues and conducting cost-benefit analyses, along with sensitivity analyses, can help assess the impact of factors like price fluctuations and cost variations on timber market income and expenses.

CRediT authorship contribution statement

Jinghan Zhao: Data curation, Formal analysis, Methodology, Writing – original draft. **William Smith:** Data curation, Formal analysis, Methodology, Writing – original draft. **Jingxin Wang:** Funding acquisition, Project administration, Supervision, Writing – review & editing. **Xufeng Zhang:** Investigation, Writing – review & editing. **Richard Bergman:** Resources, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.168458.

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