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Research article

# Spatially-explicit assessment of carbon stocks in the landscape in the southern US under different scenarios of industrial wood pellet demand

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A B S T R A C T

Whether the use of industrial wood pellets for bioenergy is part of the problem of climate change or part of the solution to climate change has been heavily debated in the academic and political arena. The uncertainty around this topic is impeded by contradicting scientific assessments of carbon impacts of wood pellet use. Spatially explicit quantification of the potential carbon impacts of increased industrial wood pellet demand, including both indirect market and land-use change effects, is required to understand potential negative impacts on carbon stored in the landscape. Studies that meet these requirements are scarce. This study assesses the impact of increased wood pellet demand on carbon stocks in the landscape in the Southern US spatially explicitly and includes the effects of demand for other wood products and land-use types. The analysis is based on IPCC calculations and highly detailed survey-based biomass data for different forest types. We compare a trend of increased wood pellet demand between 2010 and 2030 with a stable trend in wood pellet demand after 2010, thereby quantifying the impact of increased wood pellet demand on carbon stocks in the landscape. This study shows that modest increases in wood pellets demand (from 0.5 Mt in 2010 to 12.1 Mt in 2030), compared to a scenario without increase in wood pellet demand (stable demand at 0.5 Mt), may result in carbon stock gains of 103–229 Mt in the landscape in the Southern US. These carbon stock increases occur due to a reduction in natural forest loss and an increase in pine plantation area compared to a stable-demand scenario. Projected carbon impacts of changes in wood pellet demand were smaller than carbon effects of trends in the timber market. We introduce a new methodological framework to include both indirect market and land-use change effects into carbon calculations in the landscape.

## 1. Introduction

### 1.1. Wood pellet demand and supply

Whether or not the use of industrial wood pellets for bioenergy is part of the problem of, or the solution to climate change is a topic that has been heavily debated in both the academic and political arena (Ter-Mikaelian et al., 2015). While the use of wood pellets is supported by some as a way to reduce greenhouse gas emissions and reach climate change targets, others have held the production of wood pellets responsible for the destruction of forests and the contribution to the emission of greenhouse gasses (Dogwood Alliance, 2021; Greenpeace, 2021; Natural Resources Defence Council, 2020; The New York Times, 2021; The Rachel Carson Council, 2019). Negative public perceptions of bioenergy linked to these environmental risks, as well as to the local negative effects of production and power plants, has hampered the growth of the wood pellet sector (North and Pienaar, 2021; Upreti, 2004). The controversy surrounding the use of wood pellets for energy is further increased by contradicting results found by scientific studies that assess the carbon impacts of wood pellet use (reviewed by Miner et al. 2014; Booth, 2018; Cowie et al. 2021; Ter-Mikaelian et al. 2015). This study focuses on industrial wood pellets, which are brown pellets, derived from relatively low value feedstock and used by large-scale district heating and co-firing electricity installations (Goh, 2013).

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From here on, we will refer to industrial wood pellets as 'wood pellets'.

Wood pellets are easy to transport, are compatible with existing infrastructure and can be made out of residues and waste material, making them highly suitable as a fuel for energy production (Aguilar et al., 2020; Spelter and Toth, 2009). The use of solid biomass for energy production is expected to play an important role in combining a trend towards a low carbon future with sustained economic growth (Ter-Mikaelian et al., 2015; Visser et al., 2020). Several European countries have subsidized the use of solid biomass to reach renewable energy targets (European Commission, 2021), resulting in increased demand for wood pellets in Europe (Booth, 2018; Ter-Mikaelian et al., 2015). For some European countries, such as the Netherlands, the use of wood pellets is the key contributor to reaching renewable energy targets.

The EU is the largest consumer of industrial wood pellets (used for electricity production), importing 10.4 million tonnes in 2019 (out of 15.2 Mt of global demand, Gauthier et al. 2020). However, new markets are emerging in Asia; Japan and the Republic of Korea imported about 1 and 3.5 million tonnes of wood pellets respectively in 2019 (Gauthier et al., 2020; UNECE/FAO, 2020). The United States are by far the largest producer and exporter of industrial wood pellets worldwide. The US pellet producing sector showed a growth of 15% (in terms of volume) between 2019 and 2020, producing 9.5 million tonnes and exporting 6.9 million tonnes of wood pellets in 2019 (primarily to Europe) (Gauthier et al., 2020; UNECE/FAO, 2020).

About 75% of US pellets is produced in the southern region of the US (UNECE/FAO, 2020; US International Trade Commission, 2021). Here, wood is being sourced from plantation forests as well as natural forests, resulting in heated public debate in both the US and in the EU about the sustainability of wood pellets. Current debates surrounding the use of wood pellets focus on concerns about increasing wood pellet production in the Southern US, which have been raised for reasons including its potential contribution to changes in land use and forest management (Costanza et al., 2015; Prestemon and Abt, 2002) and potential subsequent negative impacts on biodiversity (Evans et al., 2013; NRDC, 2015; Olesen et al., 2016; Pelkmans et al., 2014; Tarr et al., 2016) and increased greenhouse gas emissions (Booth, 2018; Colnes et al., 2012; Junginger et al., 2019; Yassa, 2015).

### 1.2. *Carbon impacts of wood pellets*

Although the wood pellet market has grown considerably over the last decades, it still only represents <1% and <0.5% of harvested wood volume and exported forest products (by weight and value respectively) in the Southern US in 2017 (Dale et al., 2017). Nonetheless, increased demand for wood pellets has been linked to shifts in land use and forest management in the Southern US. Land use in the Southern US is dynamic, and areas of land are moving in and out of forest use over time (Miner et al., 2014; Nepal et al., 2015; Wear, 2011). About 70% of forest area in the Southern US is owned by private land owners (Hodges et al., 2019; Wear and Greis, 2013). Increased demand for wood pellets and other wood products has been projected to result in expansion of forest area in the Southern US (Abt et al., 2009; Abt and Abt, 2013; Wang et al., 2015), particularly due to an increase in the area of pine plantation (Evans et al., 2013).

A shift in land use and forest management influences the carbon stored in the landscape (Caspersen et al., 2000; Chen et al., 2006; Zhu et al., 2010), i.e. carbon stored in aboveground biomass, belowground biomass, dead organic material and soil carbon. Forests can capture and store atmospheric $CO_2$ in biomass and soil, thereby contribute to greenhouse gas mitigation (McKechnie et al., 2011). Forests in the US South stored about 12 billion tons of carbon in 2010 (Wear and Greis, 2013), constituting the largest carbon sink in the US (Chen et al., 2006). This carbon sink has been growing between 1963 and 2010, mainly due to an increase in biomass of hardwood forests (Wear and Greis, 2013). However, when forest land is cleared or wood is harvested, carbon stored in biomass and soil decreases (although not always instantaneously), which can contribute to global warming (Fargione et al., 2008).

Despite the plethora of research dedicated to the impact of wood pellets on carbon stocks and greenhouse gas emissions, uncertainty about the carbon impacts of wood pellets persists. Some academic studies that assessed the carbon impact of wood pellets found a positive impact of increased wood pellet production in the Southern US; such as an increase in carbon stocks in forests in wood pellet procurement areas (Aguilar et al., 2020; Sedjo and Tian, 2012), and savings in greenhouse gas (GHG) emissions compared to the use of coal (Galik and Abt, 2016; Wang et al., 2015). Other studies find negative impacts of wood pellets on carbon balance relative to coal-based electricity (McKechnie et al., 2014; Walker et al., 2010).

A review of carbon payback times (i.e. the period required for forest regrowth and avoided fossil GHG emissions to compensate the carbon debt created by harvesting biomass for wood pellets (Hanssen et al., 2017)) of forest bioenergy shows a range from 0 to 4500 years (Buchholz et al., 2016). Large ranges in results are in part due to differences in the system boundaries of the approach, including spatial and temporal scopes of the analyses (Dwivedi et al., 2019; Galik et al., 2015). Setting assessment boundaries provides a major challenge and can result in incomplete accounting (Buchholz et al., 2016). Elements such as market effects and indirect land-use effects (Buchholz et al., 2016; Wang et al., 2015), and their potential to result in forest expansion (Kim et al., 2018), are frequently not included in the scope of analyses. However, in the context of the Southern US, ignoring market effects disregards the main driver of historical land-use change (including natural regeneration on abandoned agricultural land and the increase of intensive forest management) and subsequent carbon stock changes. Furthermore, increased demand for wood pellets will compete for feedstock with other wood sectors and may divert feedstock from traditional wood products to wood pellets (Wang et al., 2015).

The availability and price of wood pellet feedstock are strongly influenced by developments in other large wood markets, such as the pulp & paper and timber markets (Abt et al., 2012). Low levels of harvest for more high-value products such as timber would reduce the availability of residues (Abt et al., 2012), which constitute a significant portion of wood pellet feedstock (Aguilar et al., 2020), while subsequent lower roundwood prices could stimulate the use of roundwood as a feedstock for wood pellets (Abt et al., 2012). Due to the interconnectedness of different wood markets and land uses, the assessment of the sustainability of wood pellet demand in the Southern US requires an integrated approach that includes the links between markets, land-use and environmental impacts (Aguilar et al., 2020; Cowie et al., 2021; Dale et al., 2017; Ince et al., 2011; Ter-Mikaelian et al., 2015) and assesses the carbon stored in biomass and soil throughout the landscape. Therefore, the geographical scale of impact assessments of increased wood pellet demand should extend beyond the sourcing or production area, in order to allow for the inclusion of potential indirect economic and land-use change effects of carbon stocks in the landscape. Additionally, feedstock availability for wood pellets, changes in land use, and the availability of natural resources varies strongly geographically (Cowie et al., 2021), resulting in spatial variation in the impact of land-use change on carbon stocks the landscape. Therefore, a high resolution spatial analysis of carbon impacts of increased wood pellet demand is required.

### 1.3. *Goal and objectives*

The aim of this study is to determine the impact of increased wood pellet demand on carbon stocks in the landscape in the Southern US in a spatially explicit manner, including both indirect market and land-use change effects. To this end, we quantified carbon stock changes in the landscape with and without increased wood pellet demand, and assessed the differences. We introduce a new methodological framework to include both indirect market and land-use change effects into calculations of carbon in the landscape and apply this framework to quantify the carbon stock changes in the landscape in the Southern US between 2010 and 2030 under different trends of wood pellet demand and demand for other wood products, and subsequent projections of land-use change, taken from a previous study (Duden et al., 2017). Indirect land-use change is defined as 'a change of land use outside the biofuel feedstock cultivation area, induced by a change in use or production quantity of that biofuel feedstock' (Verstegen et al., 2016). This means that, for example, as forest land competes with and replaces agricultural land (considered *direct* land-use change in this study), other land-use changes may occur where the displaced agricultural land drives agricultural expansion elsewhere, potentially displacing other land-use types such as grassland, in order to meet demand for agricultural land. We compare a scenario of increasing wood pellet demand to a scenario with no increase in wood pellet demand, to isolate the impact of wood pellet demand on carbon stocks in the landscape in the Southern US.

Summarizing, several governments promote the use of wood pellets, but have faced criticism due to sustainability concerns. Ex-ante quantification of the potential risks of increased wood pellet demand, including changes in carbon stocks in the landscape, are required to avoid negative impacts. These assessments should be spatially explicit and include both indirect effects of market interactions and land-use change in order to assess changes in carbon stocks (Kim et al., 2018). Studies that meet these requirements are scarce. Some studies include indirect market and land-use effects, but do not assess carbon impacts spatially explicitly (Abt et al., 2012; Sedjo and Tian, 2012; Wang et al., 2015), or at coarse spatial level (Galik et al., 2015), thereby restricting the possibility to supply information on potential hotspots of carbon impacts and provide input for land-use planning. This study provides a spatially explicit assessment of carbon impacts of increased wood pellet production in the landscape in the Southern US, including both indirect market and land-use effects.

## 2. Methods

### 2.1. General approach

According to a global assessment by Buchholz et al. (2016), only 12% of all reviewed studies assessing the carbon impacts of forest bioenergy were based on field data (Buchholz et al., 2016). For the Southern US, an exceptionally extensive database of detailed data on forest structure, management and tree species composition, as well as soil carbon stocks, exists in the shape of the Forest Inventory Assessment (FIA) of the USDA Forest Service (USDA Forest Service, 2020), providing an ideal opportunity for spatially explicit carbon modelling which has been used in various studies assessing carbon stock changes in forests in the Southern US (e.g see (Aguilar et al., 2022)). This study compares carbon stored in the landscape in 2010 and 2030 under different scenarios of demand for wood pellets at a resolution of 2 × 2 km, in order to assess the impact of wood pellet demand on carbon stocks in the landscape. The scenarios are based on alternative trends in the demand for wood pellets and other wood products, and consist of spatially explicit projections of land-use in 2010 and 2030 in the Southern US derived from a previous study (Duden et al., 2017).

Spatially explicit carbon values of four carbon pools (aboveground biomass, belowground biomass, dead organic matter and soil carbon) for eight land-use types (grassland, cropland, urban area and 5 different forest types) were assessed to determine the total carbon stock of the landscape in the Southern US in 2010, and for four different scenarios in 2030. We compare a trend of increased wood pellet (Aguilar et al., 2022) demand with a trend in which the increase in wood pellet demand is negligible, thereby isolating and quantifying the impact of increased wood pellet demand on the carbon that is stored in the landscape. We also include two different trends in timber demand (high increase and low increase in demand), based on assumptions on the growth of the US domestic housing market. Combined, the trends in wood pellet demand and timber demand create 4 different scenarios. We explore wood pellet-driven impacts on carbon storage by linking market projections, changes in forest area, land-use transitions and environmental impact according to the chain of driver-pressure-state-impact of the DPSIR framework (Driver-Pressure-State-Impact-Response), which was developed as a means of structuring information in a way that is meaningful to decision makers (Smeets and Weterings, 1999). More specifically, we assess 1) spatial variation in carbon storage in the Southern US in 2010; 2) potential changes in carbon storage between 2010 and 2030 in the absence of increased wood pellet demand; and 3) potential changes in carbon storage due to increased wood pellet demand. This analysis assesses carbon stocks in the landscape, but does not assess the fossil fuel substitution and life cycle impacts in terms of wood pellet greenhouse gas emissions.

### 2.2. Scenarios

The land-use projections used in this study were based on four scenarios, combining trends of wood pellet and timber demand, that run from 2010 to 2030. Two alternative trends in the demand for wood pellets were included; high demand, that assumes an increasing trend in wood pellet demand, and low demand, which assumes a stable demand for wood pellets between 2010 and 2030 in the Southern US. The High wood pellet demand trend assumes wood pellet demand to increase from 0.5 Mt in 2010 to 12.1 Mt in 2030, while the Low wood pellet demand trend assumes wood pellet demand to remain stable at the 2010 level (0.5 Mt). These trends were derived from projections of future EU (and UK) imports of wood pellets (Cocchi et al., 2011) and the proportion of imports originating from the US (Boie et al., 2016). The wood pellet market is strongly influenced by developments in the timber market (Kanieski Da Silva et al., 2019), while timber demand is driven mainly by the US domestic housing market (Ince and Nepal, 2012). Therefore, two alternative trends of the developments in the timber market were included as well. The demand for timber was assumed based on the projected development of the domestic housing market and subsequent demand for timber (Ince and Nepal, 2012), and assumes an increase in timber demand of ~90 Mt between 2010 and 2030 (high demand), or an increase of 30 Mt (low demand) between 2010 and 2030. Recent developments of the housing market roughly follow the high housing demand trajectory between 2010 and2018.[1]

Recent developments of the wood pellet market are more in line with the high wood pellet demand scenario (Figure A3c). The two wood pellet demand (high and low) and timber demand from the domestic housing market (high and low) trends were combined to create four scenarios (see Fig. 1): High housing demand & high wood pellet demand (HhHp), High housing demand & low wood pellet demand (HhLp), Low housing demand & high wood pellet demand (LhHp), and Low housing demand & low wood pellet demand (LhLp). Apart from developments in the wood pellet and timber market, scenarios also included projections of developments in other wood markets (e.g. pulp and paper) and other land uses (e.g. urbanization), these were the same for all scenarios. All four scenarios include a slowly growing demand (<2% per year) from the pulp and paper sector (based on Ince and Nepal, 2012).

Fig. 1 shows the demand for different feedstocks for wood products under the different scenarios, which was used as input into the SubRegional Timber Supply (SRTS) model (Abt et al., 2000; Galik et al., 2009). The SRTS model produces annual estimates of changes in the area of timberland required to fulfil the demand for wood feedstocks specified in our scenarios at each annual timestep at a survey-unit level. The SRTS model calculates market clearing conditions at the survey unit level, which for our study region amounts to an average area of 25,000 $km^2$ in

[1] Softwood lumber production in the US South increased by 40% between 2009 and 2018 (Greenwood, 2018) while we assume an increase in demand of 33%.

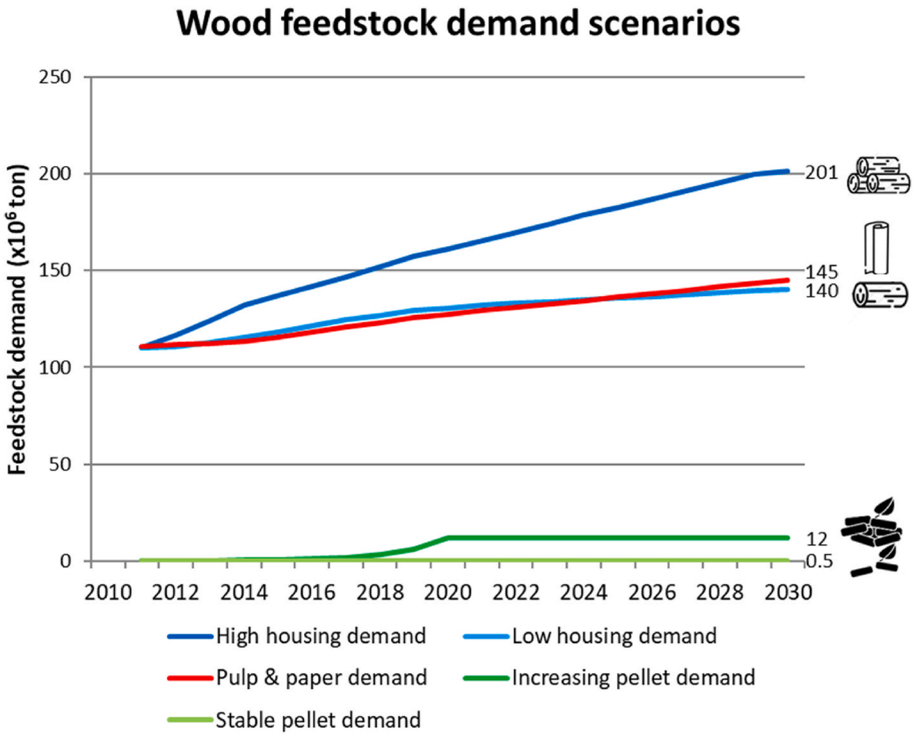


**Fig. 1.** Trends in demand from different wood markets; housing (timber), wood pellets and pulp and paper, between 2012 and 2030, used as input for the development of the four scenarios in this study. Taken from.

size. SRTS was run with default demand price, supply price and supply inventory elasticities, an annual increase in agricultural rents of 2.28% (see Figure A3d) and a transfer of 15–30% of pine and hardwood sawtimber to the pine and hardwood pulpwood pool in the form of residues (Duden et al., 2017). For trends of roundwood prices, which is an output of the SRTS model used to determine timberland area changes, see Figure A3e. SRTS determines forest area changes based on relative changes in agricultural rent and timber rents (timber rent is based on timber and pulpwood prices). Price responsiveness is assumed to differ per forest type, with pine plantations being most price responsive; Pine plantation was assumed to be 2.5 times more price responsive than other forest types. Lowland hardwood was assumed to be 0.5 times less price responsive because of hydrological and geographical restrictions in allocation. Timberland area projections from SRTS were then combined for the whole of the study area, and complemented with projections of the required area of other land uses and used as input into the PCRaster Land-Use Change (PLUC) model (Van der Hilst, Verstegen, Karssenberg and Faaij, 2012; Verstegen, Karssenberg, van der Hilst and Faaij, 2012) in order to spatially allocate changes in timberland and other land uses at a 2 × 2 km cell level. The link between the SRTS and PLUC models therefore consists of a soft link (Figure A3a). SRTS determines the changes in forest area, per forest type and per year, based on demand, inventory and price. Changes in forest area are then spatially allocated using PLUC, which is based on an allocation order and a number of land-use specific and spatially explicit suitability factors. SRTS thereby determines the rate of expansion (or loss) of forest area, while PLUC determines which land-uses are being replaced by new forest area. PLUC allocates land-use types based on required area, allocation order and land-use specific suitability factors (Van der Hilst et al., 2012; Verstegen et al., 2012).

In an earlier study, the PLUC model was adapted to the context of the Southern US by a selection of relevant land-use types and suitability factors through literature study and regression analysis of historical land use (2000–2010) with a number of spatially explicit explanatory variables (Duden et al., 2017). These suitability factors were validated for the time period 2000 to 2010 using data from the National Land Cover Database (Blackard et al., 2008). For example, the distance to wood pellet mills and other wood using mills was found to be significantly related to historical allocation of pine forests and upland hardwood forests, and was therefore included as a suitability factor in the allocation of these land-use types in PLUC. In other words, the allocation of afforestation or reforestation was, amongst other factors, influenced by the distance to wood pellet and other wood-using mills. Land-use modelling resulted in annual land-use maps at 2 × 2 km resolution between 2010 and 2030 for the following land-use types: urban, cropland, pasture, pine plantation, natural pine forest, mixed forest, upland hardwood forest, lowland hardwood forest, non-forest vegetation (which comprises all natural land-use types not classified as forest), federal land and water. Federal land, which includes protected areas, was excluded from the analysis because its land use is not expected to be influenced by market trends. Similarly, the land-use type *water* was excluded from the analysis.

In all scenarios, urbanization claimed almost 20,000 km$^2$ between 2010 and 2030 (Figure A3b; Figure A3g; Table A3a). In the HhHp scenario, pine plantation area increases by over 30,000 km$^2$, while pine plantation increases by about 25,000 km$^2$ in the HhLp and LhHp scenarios. In the LhLp scenario, pine plantation increases by over 6000 km$^2$, while natural forest area (consisting of natural pine forest, mixed forest, upland hardwood forest and lowland hardwood forest) declines by over

$14{,}000\ km^2$. The LhHp, HhLp and HhHp scenarios lose over 7,000 km,[2] over $2000\ km^2$ and almost $500\ km^2$ of natural forest respectively. Recent trends in forest area (2010–2017) for pine plantation show a stronger increase than projected by our scenarios, while for natural forests the observed decline in area falls between the high housing and low housing demand scenarios (Figure A3h). This illustrates that strong wood markets can help to keep land in forest use. Average harvest age increased during our model runs, from 34 to 39 years (see)Figure A3f, with no notable differences between scenarios. Carbon stock increases due to increased average harvest age were not included in the calculations.

The mean and sum of projected carbon stock changes between 2010 and 2030 were determined, to allow for an assessment of the carbon impact of different land-use transitions. To that end, we identified the seven main types of land-use transitions in the Southern US: *urbanization, pasture expansion, forest plantation expansion, conversion to plantation, afforestation/reforestation, forest change* and *other*.[2]

### 2.3. Carbon calculations

Based on the land use in 2010 and the projected land use under different scenarios in 2030, a spatially explicit calculation of carbon stocks and carbon stock changes in the landscape was made in order to assess the impact of wood pellet driven land-use changes on carbon stocks in the landscape. The Tier I approach of the IPCC guidelines for calculations of carbon stocks were followed (IPCC, 2006). Carbon values were determined for aboveground biomass (AGB), belowground biomass (BGB), dead organic material (DOM) and soil.

#### 2.3.1. Carbon in aboveground biomass

Carbon stocks in aboveground biomass is based on land use type, land cover type and climate zone specific data. For the forest land use types, AGB values were based on forest aboveground biomass data from the FIA. The FIA provides detailed forest inventory data on forest structure, management and tree species composition, and a full inventory cycle of over 125,000 plots located throughout the US (0ver 64,000 of which are located in the Southern US) is completed every 5–7 years. The FIA therefore provides an unusual level of detail on forest carbon, making the US an ideal study case for carbon stock calculations. Using the FIA EVALIDator tool interface (USDA Forest Service, 2020), FIA data on carbon in aboveground biomass on forest land was obtained at state level and stratified according to forest type. Aboveground biomass values for forests are based on FIA survey data and varied according to forest type (*pine plantation, natural pine forest, mixed forest, upland hardwood forest* and *lowland hardwood forest,* see Table A1b) and state (Table A1c). For urban area and for grassland, we applied IPCC default values (Table A1a). For calculations of aboveground biomass of cropland, statistics on the ratio of annual to perennial crops were determined using statistics on crop type per state from the USDA (USDA National Agricultural Statistics Service, 2019). These crop types were classified as either annual or perennial using Table A1e and equation A3 (Annex 1).

[2] *Urbanization* includes all transitions from non-urban land in 2010 to urban land in 2030. Similarly, *pasture expansion* entails a change from non-pasture land in 2010 to pasture land in 2030. *Plantation expansion* consists of a transition from non-forest land in 2010 to pine plantation in 2030, while *conversion to plantation* includes a shift from natural forest land (either natural pine forest, mixed forest, upland hardwood forest or lowland hardwood forest) in 2010 to pine plantation in 2030. Conversion to plantation will impact carbon stocks due to the shorter rotation periods of pine plantations, and subsequent lower carbon stocks. *Afforestation/reforestation* consists of a shift from non-forest in 2010 to natural forest in 2030, while *forest change* entails a shift from one natural forest type to another. The category other includes all other land-use transitions, including for example conversion from cropland or pasture to non-forest vegetation.

#### 2.3.2. Carbon in belowground biomass

Carbon stocks in belowground biomass is determined using aboveground biomass and root-to-shoot ratios. Carbon in belowground biomass was determined using equation (1):

$$C_{BGB} = C_{AGB} \bullet R \tag{1}$$

$C_{BGB}$ = Carbon in belowground biomass (tonnes/ha).
$C_{AGB}$ = Carbon in aboveground biomass (tonnes/ha).
R = Root-to-shoot ratio (dimensionless).

We used default values for R (IPCC, 2006), which are land-use type, amount of AGB, climate zone and land cover type specific. For forests, this required a crosswalk between the forest types used in this study (*pine plantation, natural pine forest, mixed forest, upland hardwood forest* and *lowland hardwood forest*) and IPCC forest types (*Tropical moist deciduous forest, Subtropical humid forest, Subtropical steppe, Subtropical mountain system, Temperate oceanic forest, Temperate continental forest – conifers, Temperate continental forest – Quercus* spp.*, Temperate continental forest – broadleaf*) (Table A1f). For cropland, R was determined based on the proportion of annual and perennial crops at state level (Table A1d).

#### 2.3.3. Carbon in dead organic material

Dead organic material was calculated using land use and forest type specific carbon stock values from the Forest Inventory Assessment (FIA) of the USDA Forest Service (Table A1h and (USDA Forest Service, 2020)).

#### 2.3.4. Soil organic carbon

Soil carbon values were obtained by using soil-type specific soil organic carbon values, which were extrapolated from inventory data of the FIA[3] (Table A1f) by Domke et al. (2017) and a land-use specific stock change factor (Table A1g), see equation (2). In the case of cropland, the stock change factor was dependent on the ratio of annual to perennial crops by state.

$$C_{soil} = D \bullet F_{LU} \bullet F_{MG} \bullet F_{I} \tag{2}$$

$C_{soil}$ = Soil organic carbon (tonnes/ha).
D = Carbon stock value per soil type and climate type (tonnes/ha).
$F_{LU}$ = stock change factor for land-use type (dimensionless).
$F_{MG}$ = stock change factor for land management regime (dimensionless).
$F_{I}$ = stock change factor for input of organic matter (dimensionless).

#### 2.3.5. Total carbon stocks

Carbon stocks were determined for 2010 and for the different scenarios in 2030. In those places where land-use change occurred, cells were assigned the a carbon value for the new land use type, based on 2010 carbon values. This ignores the fact that carbon stocks may take more than the 20 years between 2010 and 2030 to build up. This means that, for example, a newly established area of forest is assigned the forest-type and state-specific carbon value as found in the FIA database. This FIA carbon value represents an average of carbon stocks in forests of a specific type within a state, which include forests with a range of different ages. Changes in carbon stocks between 2010 and 2030, as well as between different scenarios in 2030, were calculated equation (3):

$$\Delta C_{TOT} = \Delta C_{AGB} + \Delta C_{BGB} + \Delta C_{DOM} + \Delta C_{Soil} \tag{3}$$

$\Delta C_{TOT}$ = Total carbon stock change[4] (tonnes/ha).
$\Delta C_{AGB}$ = change in carbon stocks[3] in Above Ground Biomass (tonnes/ha).

[3] The FIA only measures soil organic carbon up to a depth of 20 cm, Domke et al. have extrapolated this data to a depth of 30 cm using data from the International Soil Carbon Network (Domke et al., 2017).

[4] Change in carbon stocks between 2010 and 2030, or the difference between different scenarios in 2030.

$\Delta C_{BGB}$ = change in carbon stocks[3] in Below Ground Biomass (tonnes/ha).

$\Delta C_{Soil}$ = change in carbon stocks [3]in soils (tonnes/ha).

$\Delta C_{DOM}$ = change in carbon stocks[3] in dead organic matter (tonnes/ha).

To assess changes in carbon stocks between 2010 and 2030 in the absence of high demand for wood pellets, we subtracted the spatially variable carbon stocks in 2010 from the carbon stocks in 2030 for the HhLp scenario. To isolate the impact of increased wood pellet demand, we subtracted the carbon stocks in 2030 under the HhHp scenario from the HhLp scenario. These results are shown below. The impact of increased wood pellet demand on carbon stocks was also determined for the scenarios in which timber demand is assumed to be low (LhHp scenario - LhLp scenario), these results are shown in Annex 2.

## 3. Results

The output of the carbon stock calculations consists of maps of carbon stocks in the landscape of the Southern US for the year 2010, and for four scenarios in 2030: HhHp, HhLp, LhHp and LhLp. We explore wood pellet-driven impacts on carbon stocks by assessing 1) spatial variation in carbon stocks in the Southern US in 2010; 2) potential changes in carbon stocks between 2010 and 2030 in the absence of increased wood pellet demand (changes between 2010 and 2030 in the HhLp scenario); and 3) potential changes in carbon stocks in the landscape between 2010 and 2030 due to increased wood pellet demand (difference between HhLp and HhHp scenarios in 2030). The results for scenarios LhHp and LhLp are shown in Annex 2.

### *3.1. Carbon stock in 2010*

Total carbon stock in the Southern US is the sum of carbon in Above Ground Biomass (AGB), Below Ground Biomass (BGB), carbon in Dead Organic Material (DOM) and soil carbon, and ranges from 32 to 234 tonnes/ha in 2010 (Fig. 2b). Some hotspots occur in northern Virginia (VA), Tennessee (TN) and western North Carolina (NC). These hotspots occur in locations where dense forests occur, in conjunction with soil types of relatively high carbon content, such as inceptisols (Table A1f). Soil carbon ranges from 23 to 86 tonnes/ha. Carbon in aboveground biomass (AGB) ranges from 0 to 77 tonnes/ha, with the highest stocks occurring mainly in the northeastern regions of the study area, in the states of Virginia (VA), Tennessee (TN) and North Carolina (NC), where there is an abundance of dense upland hardwood forest (Fig. 2a). FIA data shows that pine plantation forests have relatively lower carbon stocks in aboveground biomass than the natural forest types (Figure A2a), especially in the more western and Southern states of the study area. The spatial pattern of BGB differs slightly from the spatial pattern of AGB (Figure A2f) due to the spatial differences in root-to-shoot ratios, which vary according to land-use type, climate zone and land cover type (forest type or the ratio of annual/perennial crops). Belowground biomass (BGB) ranges from 0 to 73 tonnes/ha, and is highest in the temperate states Virginia (VA), Tennessee (TN) and North Carolina (NC), because of the high aboveground biomass present in forests and a relatively high root-to-shoot ratio. Carbon in dead organic material (DOM) ranges from 0 to 22 tonnes/ha. Because DOM values are relatively high in natural pine forests, spatial patterns of high values for carbon in DOM follow the distribution of needleleaf forests .

### *3.2. Changes in carbon stocks between 2010 and 2030 in the low wood pellet demand scenario*

We assessed changes in carbon stocks between 2010 and 2030 in under a scenario of stable wood pellet demand between 2010 and 2030, but with an increasing demand for other wood products, by comparing carbon stocks in the landscape in the Southern US in 2010 with carbon stocks in 2030 in the HhLp scenario. Changes in total carbon stock between 2010 and 2030 in the HhLp scenario range from about −160 to +153 tonnes/ha (Fig. 3c). Carbon stocks increase strongest in areas that are projected to undergo afforestation/reforestation (Fig. 3b), mainly due to an increase in carbon in aboveground and belowground biomass (Fig. 3a). Decreases in carbon stocks occur mainly due to a loss of carbon in aboveground and belowground biomass as a result of urbanization, which occurs throughout the Southern US, but especially in the states of Georgia (GA), Tennessee (TN) and North Carolina (NC, Fig. 3b).

Carbon stock changes in aboveground biomass, belowground biomass and dead organic material show a similar spatial pattern (Figure A2g). The spatial pattern of changes in soil carbon differs from the pattern of AGB, BGB and DOM; Changes in soil carbon only occur in

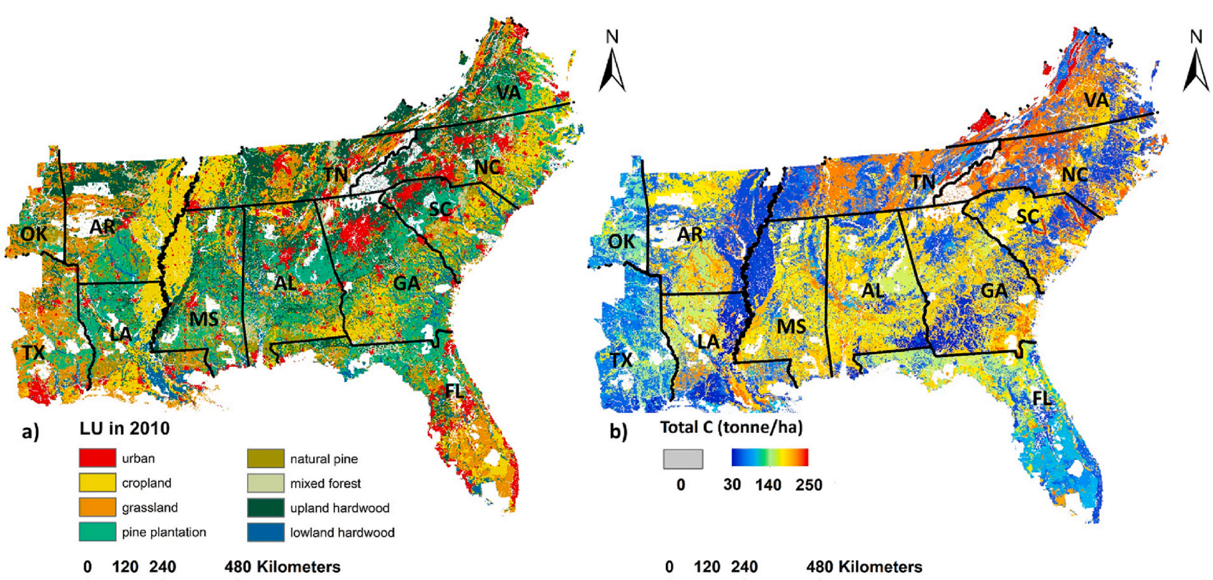


**Fig. 2.** Land use (LU) in the Southern US in 2010 (a) and total carbon stocks (tonne/ha) in 2010 (b). Black lines depict state borders. AL = Alabama, AR = Arkansas, FL = Florida, GA = Georgia, LA = Louisiana, MS = Mississippi, NC = North Carolina, OK = Oklahoma, SC = South Carolina, TN = Tennessee, TX = Texas, VA = Virginia. Carbon stocks range from 32 to 234 tonnes/ha. Gray areas have a carbon stock of 0 (figure b), white areas are excluded from the analysis (land use is water or federal land, figure a). Projection: NAD_1983_Albers.

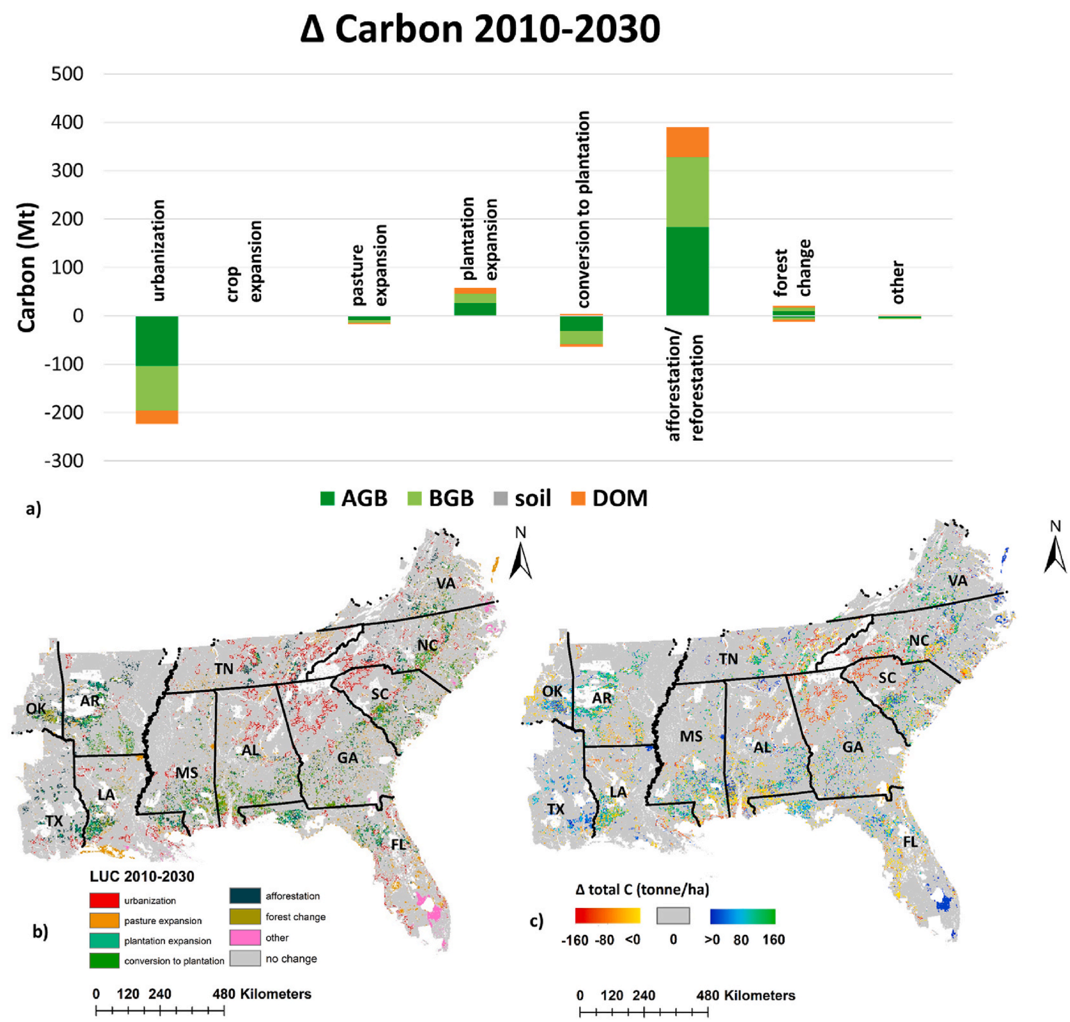


**Fig. 3.** a) Estimated carbon losses (below x-axis) and gains (above x-axis) in Mt between 2010 and 2030 in the Southern US in the high housing demand and low wood pellet demand (HhLp) scenario, i.e. in the absence of increased demand for wood pellets. Carbon losses and gains are shown per pool; aboveground biomass (AGB), belowground biomass (BGB), soil and Dead Organic Material (DOM), and per land-use change category. Between 2010 and 2030 there was no crop expansion under the HhLp scenario. b) Land use change and changes in total carbon stock (tonne/ha, b) between 2010 and 2030 under the High housing demand and low wood pellet demand (HhLp) scenario, i.e. in the absence of increased demand for wood pellets. Black lines depict state borders. c) changes in total carbon stock (tonne/ha) between 2010 and 2030 under the High housing demand and low wood pellet demand (HhLp) scenario, i.e. in the absence of increased demand for wood pellets. Black lines depict state borders. AL = Alabama, AR = Arkansas, FL = Florida, GA = Georgia, LA = Louisiana, MS = Mississippi, NC = North Carolina, OK = Oklahoma, SC = South Carolina, TN = Tennessee, TX = Texas, VA = Virginia. Gray areas have not undergone land use change (a) or have no change in carbon stock (b), white areas are excluded from the analysis (land use is water or federal land). Projection: NAD_1983_Albers.

small patches throughout the Southern US, mainly due to transitions from cropland to pasture. Across the whole Southern US, the total net carbon stock between 2010 and 2030 increases by ~165 Mt (1.1% change, Table A2a) in the absence of increased wood pellet demand (comparison of 2010 and 2030 in the HhLp scenario). When a low increase in timber demand is assumed, the net change in carbon stock between 2010 and 2030 in the absence of increased wood pellet demand is negative at −106 Mt, a change of −0.7% (comparison of 2010 and 2030 in the LhLp scenario, Annex 2).

### *3.3. Changes in carbon stocks due to increased wood pellet demand*

We assessed changes in carbon stocks in 2030 between HhLp and the HhHp scenarios, to isolate and compare the potential difference in carbon stocks in the landscape in the Southern US due to increased wood pellet demand. Differences in total carbon stock between the scenarios in 2030 range from about −160 to +160 tonne/ha (Fig. 4c). Gains in carbon stocks due to increased wood pellet demand (higher carbon stocks in 2030 in the HhHp scenario than the HhLp scenario) mainly occur due to additional afforestation and reforestation in the HhHp scenario (Fig. 4a), which predominantly results in higher carbon stocks in aboveground and belowground biomass (Fig. 4a). An additional 1800 ha of natural forest is expected to be established under the HhHp scenario (Figure A3b), predominantly in the states of Arkansas (AR) and Texas (TX, Fig. 4b). Afforestation and reforestation consisted mainly in an increase in mixed forest and natural pine forest. Decreases in carbon stocks due to increased wood pellet demand are mainly due to conversion of natural forest to pine plantation (Fig. 4a), which is projected to occur on 9900 km$^2$. Most natural forests that transitioned into pine plantation were natural pine and upland hardwood forests. This shift is strongest in the states of Georgia (GA) and Alabama (AL, Fig. 4b). Because both scenarios assume the same urban expansion, the differences between the two scenarios in gains or losses in carbon stocks due to urbanization is small. Cropland did not expand between 2010 and 2030 in scenarios HhLp and the HhHp, therefore there was no gain or loss of carbon due to crop expansion.

Summing the differences in carbon stock between the HhHp and HhLp scenarios for the entire study region, the net difference in carbon

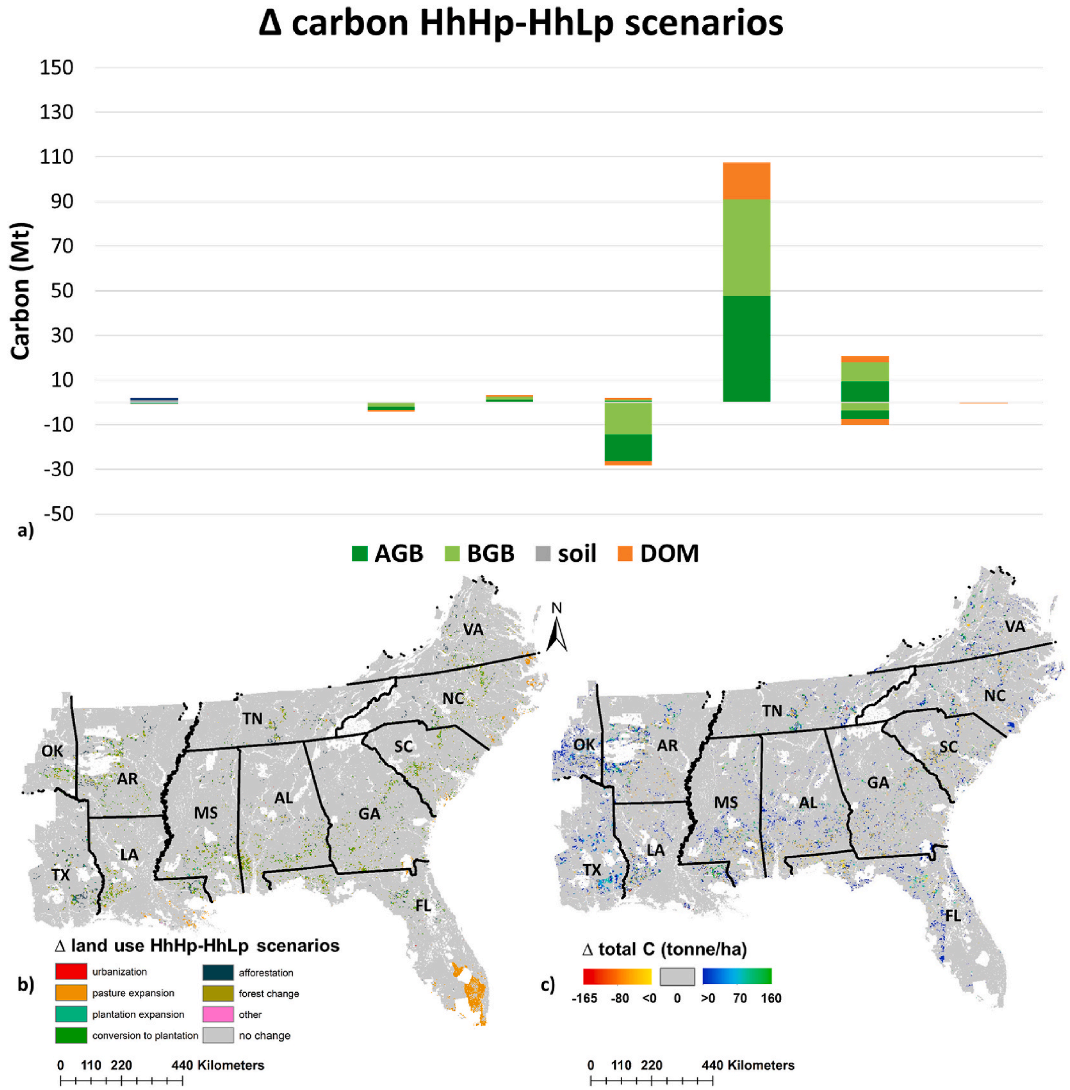


**Fig. 4.** a) Estimated carbon losses (below x-axis) and gains (above x-axis) in Mt in the southern US in 2030 between the high housing demand and low wood pellet demand (HhLp) scenario, i.e. in the absence of increased wood pellet demand, and the high housing demand and high wood pellet demand (HhHp) scenario, i.e. an increase in wood pellet demand. Differences in carbon stocks are shown per pool; aboveground biomass (AGB), belowground biomass (BGB), soil and Dead Organic Material (DOM), and per land-use change category. There was no crop expansion in 2030 when comparing the HhLp scenario to the HhHp scenario. b) Projected changes in land use in the Southern US in 2030 between the HhLp scenario, i.e. in the absence of increased wood pellet demand, and the HhHp scenario, i.e. an increase in wood pellet demand. c) carbon stocks (tonne/ha, b) in the Southern US in 2030 between the HhLp scenario, i.e. in the absence of increased wood pellet demand, and the HhHp scenario, i.e. an increase in wood pellet demand.Black lines depict state borders. AL = Alabama, AR = Arkansas, FL = Florida, GA = Georgia, LA = Louisiana, MS = Mississippi, NC = North Carolina, OK = Oklahoma, SC = South Carolina, TN = Tennessee, TX = Texas, VA = Virginia. Gray areas have not undergone land use change (a) or have no change in carbon stock (b), white areas are excluded from the analysis (land use is water or federal land). Projection: NAD_1983_Albers.

stock is 103 Mt (+0.7%), meaning that increased wood pellet demand resulted in higher net carbon stocks in the landscape. The increase in carbon stock due to increased wood pellet demand was considerably larger in absence of a strong demand for timber (difference between the LhHp and LhLp scenarios): an estimated 224 Mt (+1.6%) (Table A2a).

### 3.4. Comparison of carbon stocks in 2030 under the different scenarios

The HhLp scenario resulted in an estimated change in total carbon stocks in the landscape in the Southern US between 2010 and 2030 of 165 Mt (Fig. 5). The HhHp scenario, which includes an increase in demand for wood pellets, resulted in an increase in carbon stocks of 268 Mt. This was mainly the result of a projected increase in afforestation and reforestation, as a result of increased demand for wood products. The difference between the scenarios, caused by projected land-use changes driven by additional wood pellet demand, is ~103 Mt. The LhLp scenario, which assumes low demand for timber and no increase in wood pellet demand, results in a loss of carbon stocks in the landscape of −106 Mt. The LhHp scenario resulted in a gain of 123 Mt of carbon stocks. In the absence of strong demand for timber, therefore, the impact of increased wood pellet demand on carbon stocks was larger, and resulted in a 229 Mt difference in carbon stocks between 2010 and 2030. The difference between the different timber scenarios ranged from 145 Mt (high wood pellet demand) to 271 Mt (low wood pellet demand). The impact of different assumed trends in the development of the timber market therefore had a bigger influence on changes in carbon stocks than the variation in wood pellet demand, according to our scenarios.

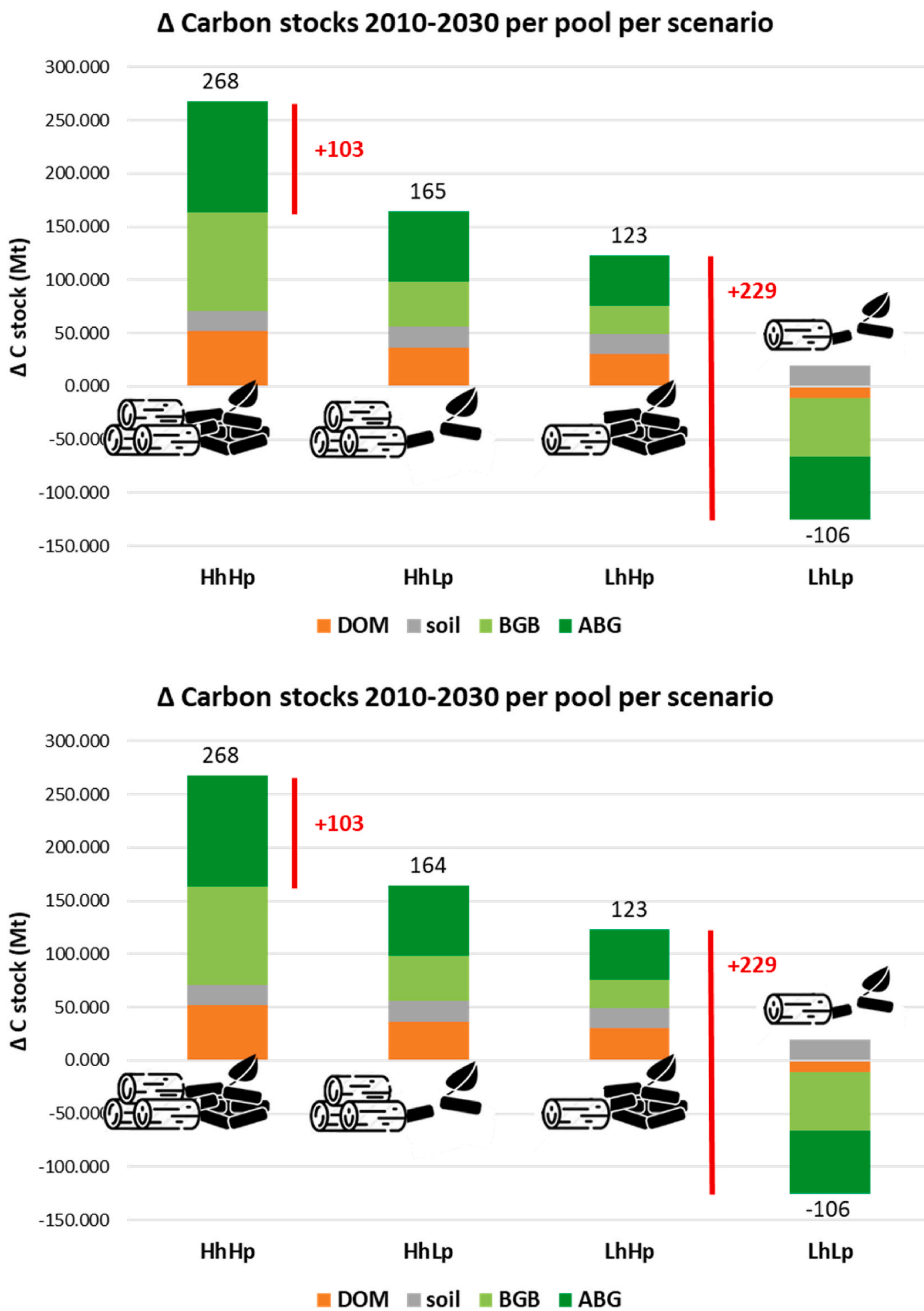


**Fig. 5.** Estimated net changes, compared to 2010, in carbon stocks (in Mt) for the different carbon pools according to the different scenarios. HhHp = high housing demand, high wood pellet demand, HhLp = high housing demand, low wood pellet demand, LhHp = low housing demand, high wood pellet demand, LhLp = low housing demand, low wood pellet demand. DOM = dead organic material, BGB = belowground biomass, AGB = aboveground biomass. Numbers show the total net change in carbon stocks between 2010 and 2030 for each scenario. Red lines and numbers show the effect of increased wood pellet demand.

## 4. Discussion

### 4.1. Main results and comparison to other studies

This study aimed to quantify the carbon impact of increased wood pellet demand, including both indirect market and land-use effects. Carbon stock changes in the landscape in the Southern US between 2010 and 2030 were determined in a spatially explicit manner for different trends of wood pellet demand and demand for other wood products, and subsequent projections of land-use change. Assuming a moderate increase in wood pellet demand, our results show an increase in carbon stock in the landscape between 2010 and 2030, mainly due to additional afforestation and reforestation. Note, however, that these stock increases may occur also after 2030. Developments in the timber market, however, were found to have a bigger positive impact on carbon stocks than these modest developments in the wood pellet market. In the absence of a strong timber market, the impact of increased wood pellet demand was still positive, but lower. Only when demand from both the timber and the wood pellet market was assumed to be low, a loss of carbon in the landscape was projected between 2010 and 2030. The carbon stocks impact due to increased wood pellet demand was projected to be +102 Mt in 2030.

We compared the results of our study with the results of previous studies. Direct comparison of our results with previous studies is complicated due to the differences in assessment boundaries. In terms of carbon budgets however, our results are in line with earlier findings. Dwivedi et al. (2019) showed that, after a break-even point, using biomass for electricity results in higher carbon stocks in the landscape (Dwivedi et al., 2019). Coulston et al. (2015) found a 6.48 Mg C $ha^{-1}$ $yr^{-1}$ increase in carbon stocks between 2007 and 2012 for changes in forest area in the Southern US making use of FIA data (Coulston et al., 2015), which is in line with our findings for the LhLp scenario – which shows an increase of 6.2 Mg C $ha^{-1}$ $yr^{-1}$. Using modelling approaches to assess the impact of demand for wood products on forest area, increased wood pellet demand was expected to result in increased carbon storage in forests of ~150 Mt in 20 years (Abt et al., 2012; Sedjo and Tian, 2012), compared to 102 Mt found in this study. A study on feedstock use and carbon flux, using a comparable scenario of wood pellet demand increase (10 Mt by 2030, but without including land-use dynamics) found little changes in carbon flux under different scenarios of wood pellet mill expansion and use of logging residues (Visser et al., 2022). In terms of spatial distribution, we found a different spatial pattern than Galik et al. (2015), who determined changes in carbon stocks between 2009 and 2029 at a survey unit level (average size of 25,000 $km^2$ in the Southern US). They identified hotspots of carbon stock changes as a result of, including other factors, shifts between agriculture and forest in eastern Louisiana, North Carolina, western Tennessee and northern Mississippi. In contrast, we found hotspots in Oklahoma and western Arkansas, as well as Texas. Our spatial analysis was at higher resolution (2 × 2 km compared to survey unit level), allowing for a more detailed impact assessment of land-use scenarios.

### 4.2. Model assumptions, limitations and uncertainties

Our results are subject to uncertainty. Validation of wood pellet production (Figure A3c), forest area changes (Figure A3h) and agricultural rent (Figure A3i) shows that our projections are in line with observed changes, apart from the observed increase in pine plantation area, which was stronger than was projected in our model. Sensitivity analyses for agricultural rent (A3j) and the assumed availability of logging and mill residues (Figure A3k) shows that our findings are robust. The scenarios for wood product demand and subsequent land use change run from 2010 to 2030. Between 2010 and the present, actual wood pellet demand has been closest to the trajectory of the "high pellet demand" scenario. The timber demand in 2018 was closer to the 'high timber demand 'than the 'low timber demand' trend. Some of the inland areas showing changes in carbon stocks do not produce wood pellets due to the distance from export ports, projected land-use change in these areas is the result of indirect land-use changes following wood pellet demand-driven expansion of natural and planted forest. These indirect land-use change effects would have been disregarded in an analysis focussing solely on the wood pellet sourcing areas, highlighting the importance of including a regional scope to capture indirect effects. Feedback mechanisms in the economic model influence results. Increased pine sawtimber consumption results in an increase in availability of mill residues, thereby offsetting pulpwood demand. We assumed that 30% of the pine sawtimber pool is transferred to the pine pulpwood pool in the form of mill residues. Especially in a scenario of high sawtimber demand and low pulpwood demand (such as the HhLp scenario), this could result in a significant proportion of wood pellet feedstock being made up out of residues. This becomes more relevant as recently, new wood pellet mills have been co-located with sawmills to increase efficiency of residue use (Drax Biomass, 2018). This recent trend shows the drive of wood pellet producers to maximize the utilization of available mill residues as wood pellet feedstock, potentially reducing the need for other feedstock types such as live trees.

The forest carbon data used in this study was based on a highly detailed field survey data provided by the FIA. Due to the size of the inventory effort, it takes 5–7 years to complete a full survey, resulting in a potential mismatch in the timing of measurements between plots. Compared to remote sensing based assessments, FIA data tends to show a lower variation in AGB estimates (Zheng et al., 2007), due to data aggregation at state level. The modelling approach used to produce the land-use projections on which this study is based has several limitations and projections should therefore not be considered as precise predictions at the pixel level, but provide an indication of potential future spatial patterns in that area. Economic modelling is used to assess the required area of forest per forest type to satisfy demand for different wood products assumes a fixed forest productivity over the modelling period. Carbon stocks are therefore not assumed to change in this study unless land use changes occur. Forest productivity could however increase over time in the presence of strong wood markets (Abt et al., 2012; Aguilar et al., 2020; Prestemon and Abt, 2002), for example due to changes in forest management related to, for instance, harvest cycle or planting densities (Jan Gerrit Geurt Jonker et al., 2018). On the other hand, forest productivity could decline due to overexploitation. Pine plantation productivity has been shown to grow by 0.5% per year on average for the US South, but with some regions showing a stronger increase (e.g. southeastern Georgia showed an increase of 2%) (USDA, 2021). Additionally, the average harvest age of forest stands increased during our model runs, but carbon impacts related to this were not included in the calculations. It has been shown that harvest age can have a significant influence on carbon balances (Dwivedi et al., 2016). However, because harvest age did not notably differ between the scenarios, we do not expect this to influence our results.

The forest area projections used in this study are based on the assumption that wood pellet demand can influence timber rents, and subsequently landowner's decisions and resulting forest area. Pine sawtimber is the primary rent driver in timberlands in the Southern US, while lower value pulpwood products, such as wood pellets, have a smaller influence on timberland rent. However, wood pellets can become more influential in determining land rents when 1) pine sawtimber prices are relatively low, or pine pulpwood prices are relatively high (or both) and 2) wood pellet demand becomes a significant share of the pulpwood market (large enough to influence pulpwood prices) (Abt et al., 2021). Both of these conditions have been met in recent years in the Southern US and were simulated using the SRTS model. Pulpwood prices have shown to be significantly increased in the Southern US around wood pellet mills (Kanieski Da Silva et al., 2019), while forestry returns relative to agricultural returns were found to be one of the main factors explaining observed timberland increase in part of the Southern US (Ahn et al., 2002; Nagubadi and Zhang, 2005). SRTS models timber rents and agricultural rents endogenously, while previous studies have shown dynamic interactions between these two sectors (Latta et al., 2013). Apart from market prices, other non-market factors have however been shown to influence landowner decision making related to land- and forest use, such as the need to raise money for health, education, or retirement (Butler et al., 2017). This was shown to be dependent on several factors including property size, and landowner age, education and income, as well as level of active or commercial management of the property (Aguilar et al., 2014; Hodges et al., 2019; Joshi and Arano, 2009; Young et al., 2015). Economic modelling assumes that private land owners are responsive to market signals, but may not capture all factors involved in landowner decision making.

The main limitation of this study is the lack of inclusion of temporal flux *within land use types* in carbon calculations. We made a simple comparison of the carbon stock in 2010 to the carbon stock in 2030 based on spatially explicit and land-use specific carbon values. This especially impacts the results related to afforestation and reforestation, by implicitly assuming that new forests reach the regional (state-specific) forest-type-specific average carbon stock by 2030. In reality, forest carbon takes time to build up, especially in the case of natural forest. Carbon recovery rate following natural forest regeneration was shown to be about 1.47 Mg C $ha^{-1}$ $yr^{-1}$ on formerly cultivated land in the Southern US (Huntington, 1995) but depends on factors including land-use history and forest age (Coulston et al., 2015). At that rate, it would take up to 50 years for new natural forests in the Southern US to reach the average carbon stock levels of existing forests (but note that these are average numbers, and therefore include both new and old growth forests). Similarly, soil organic carbon may take decades or centuries to recover from disturbance due to, for example, agricultural development (Liu et al., 2004). The rate of soil carbon sequestration following afforestation on cropland or pasture in the Southern US was shown to range between 0.1 and 0.7 Mg C $ha^{-1}$ $yr^{-1}$ (Heath et al., 2002). As a consequence, our results may provide a relatively optimistic outlook on future carbon stocks in the Southern US due to dynamics in forest area and management. Future studies may include this temporal flux within land use types, over a longer time horizon including several rotation cycles.

### *4.3. Scientific novelty and recommendations for future research*

**We introduce a new spatially explicit methodological framework to include both indirect market and land-use change effects into calculations of carbon in the landscape.** This study provides a spatially explicit assessment of carbon stock impacts of increased wood pellet demands in the Southern US to include both market effects and indirect land-use change, making use of high quality field survey-based data. Previous studies have assessed land-use changes based on timber market changes (Bigelow et al., 2022; Hardie and Parks, 1997; Lubowski et al., 2006a, 2006b; Wear, 2011). The novelty of this approach consists of the combined economic and spatial modelling, which allows for carbon impact assessment based on projected land-use transitions that take into account demand for different wood feedstocks, as well as other land uses, spatial variation in carbon stocks of different soil and forest types, and comparison of these impacts to the carbon effects of other drivers of land use change. This allows for identification of potential hotspots of change in carbon stocks at high resolution. This allows for identification of potential hotspots of change at high resolution. This study, in line with previous studies (Abt et al., 2012; Aguilar et al., 2020; Galik et al., 2015; Sedjo and Tian, 2012; Wang et al., 2015), has identified both benefits and risks of increased wood pellet production, and has shown that environmental impacts are context- and location specific.

**Future studies may want to assess more extreme scenarios of demand for wood pellets and include additional temporal dynamics.** It is highly uncertain whether the rapid growth of the Us pellets sector observed over the last decade will continue. The prime market for these pellets is industrial use in the EU, where the EU parliament has recently voted in favor of phasing out the use of whole trees (which likely includes the pulp wood used as main feedstock in US pellet mills) for power and heat. Also, national member states have cut subsidies for co-firing wood pellets in the coming years. While other markets may emerge (e.g. In the Far east), these markets may also be served by e.g. Canada or local supply. Thus, it may well be that US pellets production stabilizes around 10 million tonnes in the coming years. An assumed wood pellet demand of 12 Mt in 2030 is therefore modest and feasible, but scenarios with a more extreme growth in wood pellet demand would provide relevant insights into the relationship between wood pellet demand and carbon stocks, which may well level off and become negative as wood pellet demand increases drastically. We found a positive effect of higher wood pellet demand on carbon storage in biomass in the landscape but a more extreme increase in wood pellet demand may result in more natural forest transitioning into plantation forests, or a decline in forest productivity due to overexploitation. This could result in carbon losses in the landscape in the Southern US. Future research may aim to include a temporal flux into calculations, for example by using spatially and forest type explicit growth curves or FIA data on average annual growth per state, as well as spatially explicit assumptions or projections of forest age.

### *4.4. Policy relevance*

Ex-ante multi-model assessments, such as the analysis presented in this study, contain significant uncertainty, although sensitivity analysis and validation of input and output parameters (Figures A3c, A3h, A3i, A3j and A3k) show that our results are relatively robust. Therefore, the results should be interpreted not as a projection of future carbon stocks but rather as a way to highlight the underlying mechanisms that result in carbon stock changes. Nonetheless, some policy-relevant messages appear from our results.

**Policies that prevent the transitions of natural forests to plantations, and stimulate improved forest management will reduce carbon impacts of wood pellet production on carbon stocks.** The approach applied in this study follows a Driver-Pressure-State-Impact framework, and allows for the proposal of potential responses (the 'R' in the DPSIR framework). This study provides important information for land-use planning and shows that a modest increase in demand for wood pellets may result in carbon stock gains in the Southern US. These gains may be optimized if the projected expansion of pine plantations are directed to non-forested land, and the conversion of natural forests, particularly of carbon-rich lowland and upland hardwood forests, to pine plantations was discouraged. Keeping in mind that forest area trends are influenced by market developments and forest land is largely privately owned, a mechanism that would provide payment for carbon storage in biomass and soils may further motivate land-owners to keep their land in forest. The impact of increased wood pellet demand on carbon stocks could be reduced by increasing the productivity of forests. Southern pine plantations are among the most intensively managed forests and are highly productive (Fox et al., 2007; J. G.G. Jonker et al., 2018) but in some areas productivity may still be increased substantially. FIA data shows that carbon stocks in pine plantations were relatively high in the states of Virginia, North and South Carolina (USDA Forest Service, 2010). This could be due to favourable climatic and soil conditions, but also due to improved forest management that could be transferred to other regions in order to optimise carbon stocks. Forest management options such as increased thinning can improve harvestable wood yield while at the same time reducing GHG emissions (J. G.G. Jonker et al., 2018). Further increases in productivity could reduce the trend of increased forest planting (Abt et al., 2012), and may dampen the land-use change trends and subsequent carbon stock changes projected in this study.

**The projected land-use changes will have environmental and social impacts besides the projected impacts on carbon stocks in the landscape.** The projected land-use changes could result in a loss of forest quality – with lower biodiversity and delivery of (other) ecosystem services. The conversion of natural forests to forest plantations in the Southern US has shown to result in habitat loss for a range of specialized forest species, including species of conservation concern (Duden et al., 2018; Evans et al., 2013) and has been identified as a major risk of EU wood pellet policy (Olesen et al., 2016). An analysis of the same market and land-use dynamics shows that positive biodiversity impacts can coincide with increased carbon stocks, when afforestation occurs (Duden et al., 2018). On the other hand, the use of wood pellets, apart from reducing GHG emissions, may have other benefits, such as diversification of (energy) markets, a reduction of the dependency on fossil fuels and local socio-economic development through income and job generation (Creutzig et al., 2015; Dale et al., 2010).

### 4.5. Conclusion

Governments are faced with practical policy decisions about funding and promotion of the production and use of renewable energy. Lack of scientific knowledge or consensus on the potential climate benefits of forest bioenergy can be a significant barrier to evidence-based policy making on bioenergy issues. The approach used in this study combines economic modelling of demand scenarios, spatially explicit land-use change modelling, and spatially explicit quantification of environmental impacts in a new methodological framework. Spatially explicit modelling of potential environmental impacts, that includes indirect effects, is required to guide sustainable growth of the wood pellet sector. Assessing the (interaction) effects of different sectors is important in the Southern US, where wood markets are strongly linked and wood pellet feedstock only makes up a small proportion of the overall wood harvests. This type of approach can be applied, as shown in this study, to assess and compare the environmental impacts of alternative policy pathways related to renewable energy use.

Public perception holds that wood pellet production results in the destruction of forests. This study shows that at a regional scale in the Southern US, the opposite may occur; with moderate increases in wood pellet demand, the wood markets in the Southern US may be stimulated, motivating land-owners to shift their land use to forest or forest plantations or keep their land as forest (Wear and Greis, 2013), resulting in a reduction of the amount of natural forest lost and larger carbon stock in the landscape in the Southern US. Furthermore, the use of wood pellets, apart from reducing GHG emissions, may have other socio-economic benefits. However, the projected land-use changes could result in a loss of environmental quality in forests. Future research may address some of the limitations of this study by assessing more extreme scenarios, to understand whether the positive impact of wood pellet demand on carbon stocks in the landscape in the Southern US found in this study still holds, or could turn into a negative impact when growth in the demand for wood pellets is very strong. Some of this growth could also be absorbed through improved forest management and subsequent increases in carbon stocks per hectare. Temporal fluxes in carbon calculations could also be included into the spatially explicit assessment of the direct and indirect carbon impacts of increased wood pellet demand. Industry stakeholders may aid policy makers by providing voluntary certification systems that provides evidence of sustainable production of wood pellets. Furthermore, the spatial variability in carbon stock impacts highlights the relevance of track-and-trace systems, that provide information on the origin of wood pellets, and potentially the land-use history of wood pellet producing locations. Policy makers should keep these potential trade-offs in mind when defining regulations to safeguard sustainable expansion of the wood pellet sector in the Southern US.

## Credit_statement

**A.S Duden**: Formal analysis, writing, **P.A. Verweij**: Supervision, **A. P.C. Faaij**: Funding acquisition, supervision, **R.C. Abt:** Software, methodology, validation,**M. Junginger**: Supervision, **F. van der Hilst**: Supervision.

## 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.jenvman.2023.118148.

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