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Complete List of Authors:	Özkan, Ufuk; İzmir Kâtip Çelebi University, Dept. of Watershed Management SERENGİL, YUSUF; Istanbul University-Cerrahpasa

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A comparison of Fast-Growing Maritime Pine (*Pinus pinaster* Aiton.) Plantations with Native Broadleaved Vegetation for Greenhouse Gas Balances

Ufuk Özkan[1], Yusuf Serengil[2]*

[1]İzmir Kâtip Çelebi University, Faculty of Forestry, Dept. of Watershed Management, Izmir, Türkiye.

[2]Istanbul University Cerrahpasa Faculty of Forestry, Dept. of Watershed Management, Istanbul, Türkiye.

[*]Corresponding author (e-mail: serengil@iuc.edu.tr)

Abstract

Establishing fast-growing plantations is a long-term strategic climate change mitigation option since these plantations may absorb carbon at an accelerated rate and alleviate the pressure on natural forests. In Istanbul, nearly 5% of the forests, totaling 32,603 hectares of natural oak-beech forest ecosystems, were converted to Maritime pine (Pinus pinaster Aiton.) plantations in the 1990s. Maritime pine grows faster than native mixed broadleaf forests but introduces a higher fire risk. The objective of this study was to assess the Greenhouse Gas (GHG) consequences of these conversions by analyzing wildfire emissions and carbon stock changes for a period of 2 decades after conversion. The carbon modeling was done using the CBM-CFS3 model calibrated with ground measurements. The results revealed that the total ecosystem carbon stocks would remain at 97.9 tC/ha (Avoided Species Conversion/ASC scenario) compared to 116.7 tC/ha in the Business as Usual (BAU) scenario. The BAU scenario refers to real life conditions that the species conversions have occurred. The fire emissions had a minor share in total ecosystem GHG balance, because the burnt area rate was low (around 0.1 percent) during the assessment period.

Keywords: Fast growing plantations, species conversion, greenhouse gas balances, CBM-CFS3, carbon modelling, forest fires.

Introduction

The role of the land sector, including forestry, in offsetting anthropogenic GHG emissions is underlined frequently (IPCC 2019; Cintas et al. 2017; Majava et al. 2022) . The sector is both a significant source of emissions and removals. The emissions caused by the disastrous forest fires in Türkiye in 2021 were 10.46 Mt CO2 eq, while removals in the same year accounted for 35.10 Mt CO2 eq according to the 2023 National Inventory Report [NIR Turkey, 2023]. The contribution of forestry to national GHG removals is achieved mainly through forest management (FM) and afforestation/reforestation (AR) activities (Serengil, 2018). Forest management involves several forestry treatments and activities occurring in forestlands, while AR defines a land conversion to the forest. Species conversion or planting of fast-growing trees in place of native forests is technically a forest management treatment [Roth et al. 2023] with substantial impacts on ecosystem functions and dynamics, including biodiversity and water balances. In case of species conversion by fast-growing species, the ecosystem regenerates and grows with new dynamics, resulting in changes in biomass and net primary production (i.e., Heine et al. 2019; Desie et al. 2019). The dead organic matter (DOM) and soil responses come slower, driven by the site conditions and species used. Consequently, the gradual changes in forest stand composition due to conversion alter DOM carbon by changing accumulation and decomposition dynamics (Desie et al. 2019).

Species conversion to establish fast-growing plantations has been one of the widespread forestry practices for decades [Xu and Mola-Yudego 2021; Smyth 2023]. The primary motivation for conversions was to increase the quantity and quality of timber production in relatively short periods, while the concerns were their low resiliency to disturbances in new ecological conditions [Gaspar et al. 2020; Díaz et al. 2024] and, in some cases, their invasiveness [van Etten et al. 2020]. However, most plantations survived and successfully increased aboveground net primary production compared to native ecosystems (Araujo and Austin 2020).

The growth and associated carbon removal rates of fast-growing species may differ significantly among them and as compared to native vegetation based on ecological conditions and disturbances. The water transport systems and utilization patterns differ in fast-growing exotic and native tree species [Ni et al. 2024]. In low annual precipitation conditions (below 650 mm), carbon accumulation in aboveground biomass was much higher, and decomposition rates were lower in exotic *Pinus ponderosa* Douglas ex P. Lawson & C. Lawson plantations than in the native forest. The difference in carbon stocks (C stocks) between the plantations and the native forest decreased with increasing precipitation [Araujo and Austin, 2020]. The short- and long-term water deficit also affects the carbon balance in maritime pine plantations. Even seasonal droughts may reduce growth and cause the ecosystem to become a net GHG emitter or reduce the removal rate significantly [Jarosz et al. 2008].

On the other hand, conversions to fast-growing conifers may exacerbate the risk of forest fires that may cause GHG emissions [Jimu and Nyakudya, 2018; Labbé et al. 2023; Lindenmayer et al. 2023]. The risk can be reduced to some extent by certain fuel management measures [Hevia et al. 2018], such as controlled burning, mechanical shredding, and thinning. These treatments performed at critical spots affect the fire occurrence and spread significantly but do not affect the whole ecosystem health and functionality [Plaza-Álvarez et al., 2024; Hevia et al., 2018]. No vegetation control measure was implemented in our case, and even traditional thinning was not done properly. Therefore, the fire risk was higher than that of native vegetation.

In Istanbul, the conversion of native broadleaf forests to exotic pine stands in the late 1990s corresponded to a period of city expansion and increased human activities in or around forests. The period is characterized by increased human pressure on forests resulting in increased wildfire occurrences. Therefore, the trade-off between increased wood production and fire emissions has drawn the attention of authorities and policymakers.

This study aims to quantify and evaluate the trade-off between enhanced growth and increased fire occurrences due to the 1990s species conversions in the Istanbul region from a GHG balance perspective. We tested whether increased fire emissions have offset the carbon removal benefits of fast-growing exotic species. A carbon dynamics model (CBM-CFS3) was calibrated with field measurements to calculate the long-term net GHG balance.

Materials and Methods

Study Area

The study was conducted in the Istanbul Forest Region in northwest Turkey (refer to Figure 1A). Over the past few decades, the forests in the northeast and northwest regions (refer to Figure 1B) have been reducing in size, especially after the 1980s due to the country's economic growth. The loss of forest connectivity occurred due to the consumption of croplands and forests by urban areas and the construction of new roads and infrastructure. Landscape fragmentation and management prevented wildfires from spreading and becoming disastrous in the native deciduous forests.

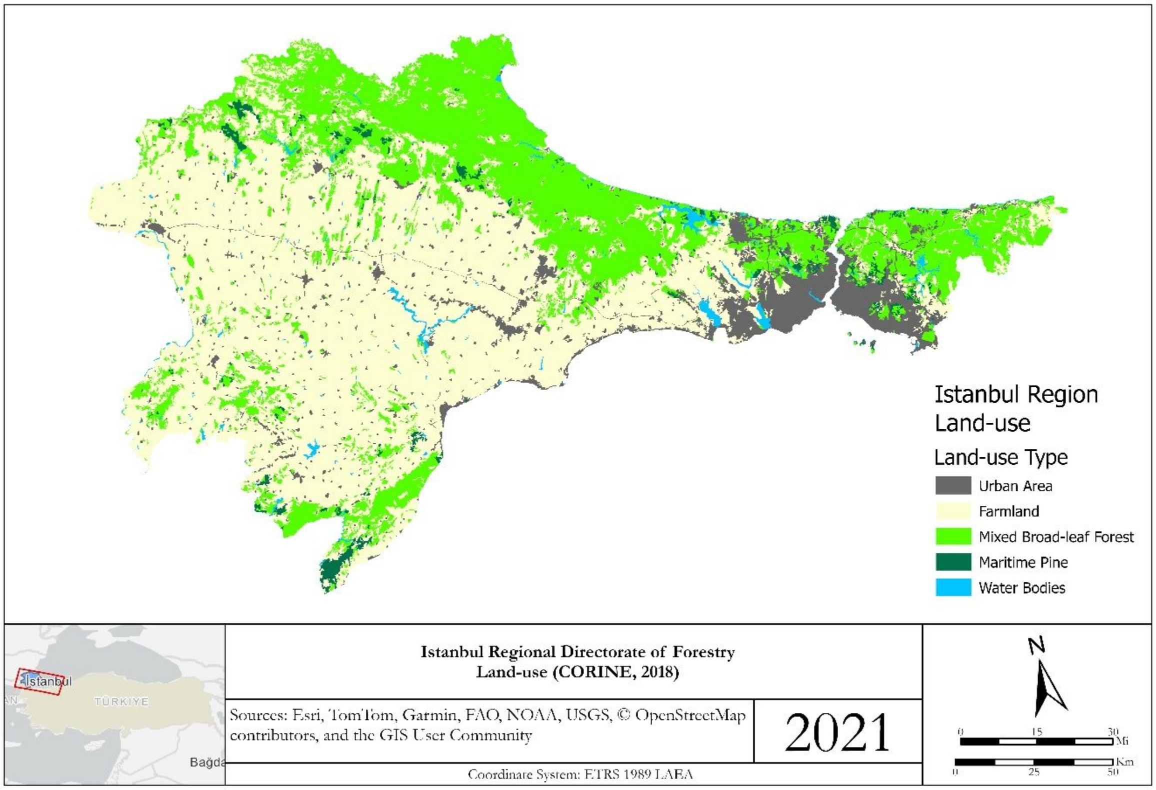


Fig. 1. Location of the Istanbul Forest Region and Land use. Land-use Data: European Environment Agency. (2018). CORINE land cover 2018: Land use change map [Data set]. Copernicus Programme. Retrieved from https://land.copernicus.eu/ (Open source).

The region's climate is notably influenced by the nearby sea, giving way to a temperate environment. An average of 662.5 mm of precipitation annually and a mean temperature of 15.3 °C characterize the area (Table 1). As concluded by Serengil (2018), the forest ecosystems in this region align with the Balkan Mixed Forests and Euxine-Colchic Broadleaf Forests eco-zones.

Table 1. Long-term (1950-2023) climatologic data for the study area (mgm.gov.tr).

	Ja	Fe	Ma	Ap	My	Jn	Jl	Au	Se	Oc	No	De	Annual
Tmean (°C)	6,7	6,9	8,4	12,8	17,6	22,2	24,6	24,7	21,2	16,7	12,6	8,9	15.3
Tmax (°C)	9,6	10,2	12,3	17,3	22,2	26,9	29,6	29,6	25,9	20,6	16,0	11,8	19.3
Tmin (°C)	4,2	4,2	5,4	9,2	13,6	18,0	20,4	20,7	17,6	13,7	9,8	6,4	11.9
Insolation (hr)	0,8	0,4	1,0	1,3	1,1	1,1	1,1	1,4	1,0	0,2	0,5	0,6	0.9
Rainy days (mean)	16,5	14,2	12,8	10,3	7,7	5,6	3,6	3,6	5, 6	9, 6	11,5	15,7	116.5
Pmean (mm)	88,6	71,3	63,3	48,5	32,8	27,9	22,2	24,3	40,0	66,2	78,8	98,6	662.5

The forest cover in the Istanbul Region spans over 638,000 hectares, with a predominant composition of mixed native broadleaved species. In 1997, fast-growing exotic Maritime pine (*Pinus pinaster* Aiton.) plantations were introduced into the region as a replacement for the native forests at some certain patches. According to the national forest inventory (Inventory Information System - ENVANIS), around 67,000 hectares were planted, while 32,603 hectares remained until today.

Methodology

The methodology is based on two basic scenarios, namely Business as Usual (BAU) and Avoided Species Conversion (ASC), to simulate and compare the dynamics of GHG balances. The BAU scenario was based on actual burnt area data obtained from the Forest Service, while the ASC scenario was computed using a reduction coefficient explained below.

Forest Fire Data

Tabular forest fire data of the Istanbul Forestry Directorate for 2001-2018 included date and time, location, burnt area, forest type, tree species, and cause of the fire. The database is further digitized and linked with forest inventory data, which is used to estimate fire emission rates for the scenarios.

The observed burnt area data provided by the Forestry Directorate were used to calculate historical GHG emissions in the BAU scenario, while the burnt areas for the ASC scenario was estimated using the ratio between the proportions of the total burned and forested areas for the two forest types in the 2001-2018 period. This assumption reduced the area burned in the no-conversion scenario and the associated emissions.

Consequently, forest fire emissions were simulated using two scenarios. The first scenario (BAU) used the actual area of pine stands burned. The real data for native and converted stands were used in GHG emission calculations. In the second scenario (ASC) the natural mixed broadleaf stands would assumed to remain unconverted. In this case, there was no conversion to Maritime pine; all forests were native broadleaves. Thus, the burnt area calculated was less. The proportion of each forest type that burned in any year between 2001 and 2018 has been estimated as,

$$Proportion\ of\ forest\ type\ burned\ in\ year\ n = \frac{Burnt\ area\ of\ the\ forest\ type\ in\ year\ n}{Total\ area\ burnt\ in\ year\ n}$$

The ratio of the proportion of area burned in the two forest types have been calculated as,

$$Ratio\ of\ ASC = \frac{Proportion\ of\ mixed\ broadleaf\ stands}{Proportion\ of\ Maritime\ pine\ stands}$$

The estimated area to be burned in case of no conversion (ASC scenario) is calculated by multiplying the burnt area of maritime pine in year n with the ratio of ASC. The estimated area burnt will be significantly reduced because the proportion of burned broadleaf forest is relatively lower than the Maritime pine stands.

Forest Inventory Data

Türkiye's Forest Inventory System (ENVANIS) is the main data source for the Istanbul Forest region. The system is not a typical forest inventory. It is based on intensive temporary sampling plots to produce Forest Management Plans. The inventory database included stand age, annual volume increment, main tree species, last and historical events, current stock, harvest schedule, and disturbances.

Field Sampling and Analysis

A field sampling was performed to calibrate the CBM model. The aboveground biomass and soil sampling was done at typical circular forest inventory plots of 400 m2 each. In the plots, the diameter at breast height and tree height were measured for all individual trees to estimate aboveground biomass using species-specific allometric equations. Litter sampling was also done at four randomly selected points of the plots using 0.5 m x 0.5 m quadrats. Double litter samples were dried and weighed in the laboratory. Standard double-bulk soil samples were taken at each quadrat to estimate the soil organic matter. Carbon stocks were calculated by using carbon fractions specific to each carbon pool. Twelve of the 89 sampled plots were Maritime pine stands; the rest were native broadleaved forests. The number of plots for maritime pine plantations were lower than native broadleaves. However, the diameters and heights were relatively uniform to derive the stand parameters needed for the model. The actual carbon stock data were used to calibrate and validate the CBM model parameters.

CBM-CFS3 model

The carbon dynamics for our scenarios were evaluated with the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) that represents carbon dynamics in the five carbon pools, aboveground biomass, below-ground biomass, litter, woody debris, and soil carbon (Kurz et al. 2009). The model parameters and disturbance matrices for management practices were described by many previous studies (i.e., Kurz et al. 2009; Smyth et al. 2011).

The CBM-CFS3 model (Kurz et al. 2009) is a well-established and documented model to estimate carbon stocks and stock changes in forest ecosystems. Region-specific parameters and inventory data are used to estimate carbon dynamics in forest carbon pools following Intergovernmental Panel on Climate Change (IPCC) standards. The model has been used in Canada and around the world to assess carbon balances (Stinson et al. 2011; Kim et al. 2016; Pilli et al. 2013; Pilli et al. 2017) and mitigation options (Smyth et al. 2016; Olguin et al. 2018; Dugan et al. 2018).

The model can simulate the changes in all carbon pools in a forest ecosystem consistent with Tier 3 methods (IPCC 2006). It is particularly useful in precisely estimating carbon balances in case of disturbances. The required input parameters are the tree species, growth and yield curves, activity and disturbance data, a forest harvest schedule, and land-use change information. The model uses ecological parameters and annual mean temperature to calculate litter, dead wood, and soil organic carbon stocks and stock changes. It can be applied at different scales (stand, landscape, regional, or national). Additionally, with the appropriate input data, CBM-CFS3 can simulate past C dynamics and project future dynamics under different scenarios.

The CBM-CFS3 is used to estimate and report GHG emissions and removals in managed forests of Canada (Stinson et al. 2011; Kurz et al. 2018) and is applied in other countries for verification purposes (Pilli et al. 2017). The model uses the IPCC Gain-Loss approach to estimate past and projected future carbon stock changes in annual time steps, considering forest management, natural disturbances, and land-use change (Kurz et al. 2009; Kurz et al. 1992; Kurz et al. 2018). In this study, the model was mainly used to estimate carbon removals in forest ecosystems and the emissions from forest fires.

The CBM-CFS3 uses empirical yield curves to estimate aboveground biomass dynamics. Our growth curves were developed based on age-volume relations for forests in the Istanbul Region from forest inventory tables. Stand ages were determined by dividing total volume by the average annual increment, with increment values provided by the General Directorate of Forestry. Based on these calculated stand ages, we identified the most common stand age classes by species and calculated an average volume for each age class. Ages and volumes for each stand were listed, and growth curves were fitted using the Chapman-Richards growth function (Zhao-gang et al. 2003; Pienaar and Turnbull 1973; Kuželka et al. 2015) for Maritime Pine and Mixed Broadleaved forests (Figure 2). MATLAB-based KORFit add-on was used to create the growth curves.

$Y(t)=a \cdot (1-e^{-b \cdot t})^{c}$

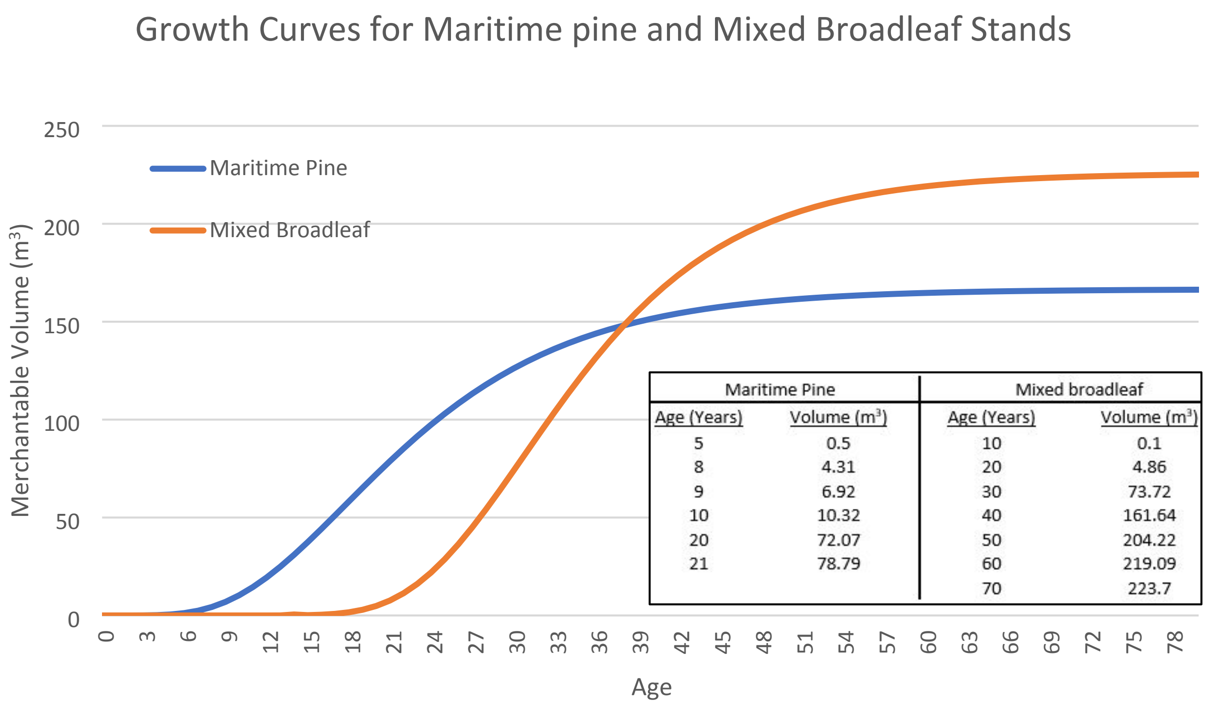
$Y(t)$ is the predicted size (biomass, volume, etc.) at age t,
a, b, and c are parameters to be estimated.

For Maritime pine;

a=166.57
b=0.1058
c=6.52c

For Mixed broadleaf;

a=225.78
b=0.1199
c=40.31



Maritime Pine		Mixed broadleaf	
Age (Years)	Volume (m^3)	Age (Years)	Volume (m^3)
5	0.5	10	0.1
8	4.31	20	4.86
9	6.92	30	73.72
10	10.32	40	161.64
20	72.07	50	204.22
21	78.79	60	219.09
		70	223.7

Fig. 2. Growth curves for Maritime Pine and Mixed Broadleaf species derived from the national forest inventory in units of merchantable stem volume (of trees greater than 8 cm dbh).

The data used in the calculations following the IPCC (2006) guidelines are supported by national or local research and complemented by data collected from ENVANIS (Türkiye National Forest Inventory-Statistics). Since detailed stand growth curves for each area are not available for modeling in the CBM-CFS3 software, unique stand growth models were created from ENVANIS, and an annual increase and total volume estimates of each forest stand for a particular year were made. Growth curves to be used in CBM-CFS3 have been explicitly developed for hardwood and softwood species via KorFit, and volume/age values have been taken as a basis while developing.

CBM-CFS3 additionally requires data (Table 2) on the process, like annual temperature, tree species growth patterns, decomposition coefficients, and other parameters to simulate carbon pool transitions, releases, and sequestrations (Kull at al., 2017).

Table 2. The stand-level Project Creator processes (Kull, 2019).

Step 1	**Project basic specification (Region, ecozone, forest type/species, and soil type)**
Step 2	Project optional specification (Stand number, simulation period, growth curve informa
Step 3	Define stand attributes (Stand definition like age, area, species)
Step 4	Define disturbance types (Disturbance or forest management activities types and their
Step 5	Schedule disturbance events (iteration and interval information)
Step 6	Enter growth and yield information (input growth curves or yield models)
Step 7	Modify project parameters (optional)

The CBM-CFS3 quantifies the impacts of natural and human-caused disturbances on ecosystem carbon stocks using disturbance matrices defining each disturbance type and intensity. The proportion of existing ecosystem C stocks that remain in their current pool is transferred to another pool (e.g., merchantable biomass C to standing dead) or the atmosphere (as CO_2, CO or CH_4) or the forest products sector (Kurz et al. 1992; Kurz et al. 2009; Kull et al. 2019). Default disturbance matrices of clear-cut harvest and wildfire as defined in the CBM-CFS3 "Archive Index Database" compiled for CBM-CFS3 applications in EU countries were used (Pilli et al. 2016b). A specific region with similar mean annual temperature and total annual precipitation conditions was chosen in the EU database. In addition, mean annual temperature data for the Istanbul Region were provided as input to the model.

Archive Index Database for European Union countries (open source), containing regions ecologically similar to Türkiye. The closest region, Sud-Vest Oltenia, with Ecological boundary CLU36 (RO41) to Istanbul, was selected. The selected species were hardwood and softwood, while the soil type was average (Pilli et al., 2018). The annual average temperature was 12.5 °C and the total precipitation was 750 mm (the model run was modified with Istanbul data). Except for climatic data, species type and ecoboundry defaults were used. Soil carbon stock value directly taken from the National Inventory Report (NIR) of Türkiye was 70.63 tonnes C/ha, and decay rates were taken from EU database (Table 3).

Table 3. The default coefficients used in the CBM-CF3 model run specifically for the forest type and ecologic region.

Input parameter	Value	Input parameter	Value	Input parameter	Value
AverageAge	150	bark_a	-2.07	softwoodFoliageFallRate	0.11
def_spuid	12	bark_b	-0.00052	hardwoodFoliageFallRate	0.95
def_species_type	145	bark_c	0.14	stemAnnualTurnOverRate	0.003
merch_a	1.39	branch_a	-2.98	softwoodBranchTurnOverRate	0.04
merch_b	0.82	branch_b	-0.00173	hardwoodBranchTurnOverRate	0.04
non_merch_a	4.77	branch_c	0.03	averageDOM	0
non_merch_b	-0.97	vol_min	16.7	decayMult	1
non_merch_c	1	vol_max	266.65	softwoodStemSnagToDOM	0.032
non_merch_cap	1.41	pstemwood_low	0.71	hardwoodStemSnagToDOM	0.032
sapling_a	1	pstemwood_high	0.69	softwoodBranchSnagToDOM	0.1
sapling_b	0	pstembark_low	0.11	hardwoodBranchSnagToDOM	0.1
sapling_c	0	pstembark_high	0.1	soil C stock	70.63
sapling_cap	1	pstembranch_low	0.13		
stem_a	-1.94	pstembranch_high	0.16		
stem_b	-0.00044	pfoliage_low	0.03		
stem_c	0.02	pfoliage_high	0.02		

We calculated the HWP pool with country-specific coefficients and EFs using the First Order Decay Function (FODF) as IPCC (2006) suggested. The FODF (2) and the parameters we used were:

$$C(i+1)=e-k*C(i)+((1-e-k)/k)*Inflow(i) \qquad (2)$$

Where:

i = year,

C(i) = the carbon stock in the particular HWP category at the beginning of the year i, Gg C, k - decay constant of FOD for each HWP category,

k – ln(2)/HL, where HL is the half-life

Inflow(i) - the inflow to the particular HWP category during the year,

(i) - Gg C. yr−1.

The carbon stock change of the HWP category during the year (i) is calculated (3);

$$\Delta C(i)=C(i+1)-C(i)” \quad (3)$$

The half-life value for sawn wood is 35 years,

The half-life value for wood panels is 25 years,

The decay constant of FOD for sawn wood is 0.020 and

The decay constant of FOD for wood panels is 0.028.

Scenario Assumptions

Species conversion aims to produce high-quality industrial wood in less time. In 1997, the process of converting tree species began in the region by planting one-year-old Maritime pine seedlings after clear-cut harvests and soil preparation. These plantation patches were scattered throughout the region using seedlings imported from Italy. The simulations were limited to the 32,603 hectares of forests converted from native deciduous broadleaf species to Maritime pine.

The CBM simulation involved two scenarios - Business-as-usual (BAU) representing the actual historical sequence of species conversions and Avoided Species Conversion (ASC) representing conditions if natural mixed broadleaved stands remained unconverted. The simulations included wildfire events, followed by salvage logging and replanting with saplings older than one year with the same species. The scenarios were simulated from 1990 to the end of 2018, with the BAU scenario using maritime pine and the ASC scenario replanting with broadleaf species (Table 4).

Table 4. The details of the scenarios simulated with CBM-CF3 model.

Scenario	Scenario 1- BAU	Scenario 2- ASC
Description	The forest patches were converted to Maritime pine in 1997	The forest patches were regenerated with native deciduous forest cover in 1997
Management	The seedlings were planted at a specific distance, and complementary plantations were made in the first year. No silvicultural treatments have been until 2018. The rotation period for Maritime pine in Türkiye is 30-50 years [Karakuyu and Ozçelik, 2020; Kocabıyık et al., 2023].	The seedlings were planted at a specific distance, and complementary plantations were made in the first year. Traditional release cuttings were performed in the early ages to open space for the individuals. The rotation period of mixed broadleaves is 100-120 years [Şahin, 2023].
Area (ha)	32,603	32,603
Site preparation	Shallow mechanized soil preparation, shrubs and herbs removed	Shallow mechanized soil preparation, shrubs and herbs removed
Fire area assumption	All actual fires occurred in the simulation period	A reduced burnt area was used to reflect a lower fire risk of native vegetation (Tab. 3).
Post fire treatment	Salvage logging followed by planting 1-yr old seedlings	Salvage logging followed by planting 1-yr old saplings
Carbon pools excluded	The litter pool also covered the dead wood.	The litter pool also covered the dead wood.
Fire emission areal data used	1213 ha burnt forest (747 ha native broadleaved + 456 ha maritime pine) (Real data from Forest Service records)	783.7 ha burnt forest (747 ha + 36.7 ha) broadleaved burnt (Real data for native broadleaved plus adjusted burnt area in case of no conversion)

The percentage of burnt areas in both maritime pine and native deciduous forests between 2001 and 2018 were analyzed, and found that if the region had remained as native deciduous forests (ASC), the burnt area would have been significantly reduced. The proportion of burnt native deciduous forests to the proportion of burnt planted pine forests during that period was calculated to estimate the burnt area in the ASC scenario. The emissions for the ASC scenario were then estimated using this recalculated area.

Consequently, the removals were calculated for 32 603 ha for both scenarios but burnt areas to calculate emission were different. In the BAU scenario, burnt area for the 2001-2018 period was 1203 ha (747 ha native broadleaved plus 456 ha martime) while it was 783.7 ha (747 + 36.7 ha native broadleaved). The reason for the reduced burnt area is the around ten times lower burnt area of native broadleaved as compared to maritime pine recorded in the 2001-2018 period. The calculations are explained in Table 5 below.

Results and Discussions

In this study, the GHG balances of two forest types from plantation to middle ages were compared. While the fast-growing plantations stocked faster, they also caused higher wildfire GHG emissions.

Over the period 2001 to 2018, fires in the Istanbul Forest region occurred predominantly near urban areas with much lower frequency in rural areas (Figure 3). The number of fires was variable, slightly increasing during the last decade. The fires were generally small to medium scale and involved a few hectares. The major limitation of spreading was the fragmentation of forests. Moreover, the native vegetation of the region is mixed broadleaved forests that are less prone to burning.

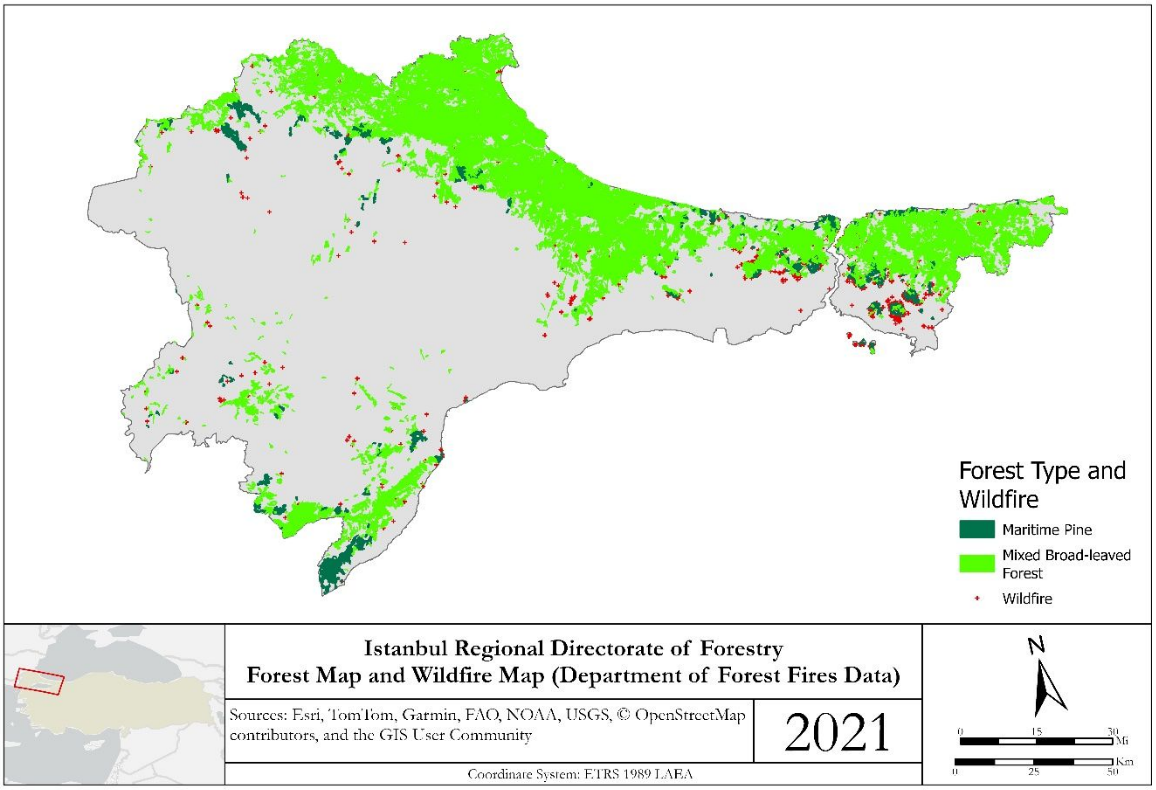


Fig. 3. Istanbul Region Forested Areas and Forest Fire Events from 2001 to 2018 (Data from the General Directorate of Forestry). The fires in farmlands are controlled biomass burning. Wildfire Data: Istanbul Regional Directorate of Forestry, Department of Forest Fire. (n.d.). Fire data [Data set]. Access to this data is restricted and requires special permission.

The fire events for Maritime pine and mixed broadleaved stands (Table 5) show that from 2001 to 2018, 456 ha of pine and 747 ha of broadleaved forests were burned. Fires affected 1.4% of the 32,603 ha of Maritime pine and 0.12% of the broadleaved forests. The burnt area of maritime pine stands fluctuated between zero and 143 ha (2004). On the other hand, even though the maritime pine stands covered only 5 percent of the total regional forests, they outnumbered the broadleaved stands in the number of fires. Therefore, based on the period's statistics, broadleaved stands were more than ten times less prone to wildfires, and fire frequency has been much less than maritime pine. However, despite the higher susceptibility to fire, the total burnt area for the maritime stands remained at 1.4% over the period.

Table 5. Total burnt area and number of forest fire events for total (Istanbul Region), Maritime pine, and broadleaved stands between 2001 and 2018. The percentages are also given in parentheses. The bold numbers are the activity data used to calculate fire GHG emissions.

Years	Istanbul Region		Maritime Pine Stands				Mixed Broadleaved Stands				
	Total Burnt Area (ha)	Total Number of Forest Fires (n)	Total Burnt Area (ha)(%)		Number of Forest Fires (n) (%)		Total Burnt Area (ha)(%)		Number of Forest Fires (n) (%)		Adjusted Burnt Area* (ha)
2001	239	170	**15**	(6.3)	52	(30.6)	171	(71.5)	72	(42.4)	**1.2**
2002	31	50	**7**	(22.5)	14	(28)	22	(70.5)	28	(56)	**0.6**
2003	204	204	**43**	(21.2)	103	(50.5)	98	(48.1)	49	(24)	**3.5**
2004	145	68	**143**	(99.2)	66	(97.1)	0	(0.1)	0	(0)	**11.5**
2005	32	58	**4**	(12.9)	21	(36.2)	19	(59.8)	19	(32.8)	**0.3**
2006	68	150	**13**	(19.6)	65	(43.3)	30	(44.2)	54	(36)	**1.0**
2007	264	186	**74**	(28)	91	(48.9)	164	(62.1)	54	(29)	**6.0**
2008	96	137	**7**	(7.2)	55	(40.1)	27	(27.6)	36	(26.3)	**0.6**
2009	92	161	**41**	(44.8)	63	(47.7)	37	(40.6)	98	(27.7)	**3.3**
2010	7	42	**0**	(4.5)	15	(35.7)	6	(80.4)	15	(35.7)	**0.0**
2011	68	171	**9**	(13.4)	67	(39.2)	6	(8.6)	43	(25.1)	**0.7**
2012	104	230	**22**	(21.3)	83	(36.1)	52	(50.3)	69	(30)	**1.8**
2013	77	271	**10**	(13.3)	114	(42.1)	34	(44.8)	61	(22.5)	**0.8**
2014	18	108	**4**	(23.6)	65	(60.2)	10	(51.8)	20	(18.5)	**0.3**
2015	41	153	**12**	(27.9)	76	(49.7)	16	(39.6)	36	(23.5)	**1.0**
2016	92	254	**26**	(27.8)	129	(50.8)	33	(36.1)	61	(24)	**2.1**
2017	39	170	**12**	(31)	80	(47.1)	14	(36.1)	38	(22.4)	**1.0**
2018	25	80	**14**	(56.3)	50	(62.5)	8	(31.6)	12	(15)	**1.1**
Total	1642	2663	**456**	27.77%	1209	45.40%	747	45.49%	765	28.72%	**36.7**

- *Recorded burnt area has been multiplied by the "Ratio of ASC" given in the methods section.

We had actual burnt area data for the BAU scenario to estimate the burnt area of the ASC scenario as explained in the methods section.

The ratio to estimate the burnt area for the ASC scenario was 0.0804. We calculated the burnt area for the ASC scenario by multiplying the actual burnt area with the coefficient calculated. The adjusted burnt area, in this case, was reduced more than tenfold as native broadleaved forests were recorded to burn less in the given period.

The largest forest fire in Maritime pine stands was in 2004 and covered an area of 143 ha. The burnt area was 0.44 percent of the total area. The average annual burnt area during the assessment period was usually around 0.1 percent of the total forested area.

Emissions in the BAU scenario were considerably higher than in the ASC scenario since Maritime pine stands had larger burnt areas (Figure 4). In 2004 the fires caused more than 1550 tonnes of CO2 emissions in the BAU scenario while the emissions in the ASC scenario remained at 136 tonnes of CO2. However, in both scenarios, fire emissions did not cause any significant decrease in the carbon balance of the ecosystems since the burnt areas were tiny compared to the whole forest area during the simulation period. The ratio of the burnt areas did not get over 0.21 percent during the period and oscillated around 0.1 percent. The graphs also show the high interannual variability of forest fire emissions.

Consequently, the annual wildfire emissions were below 1550 tCO2 eq/yr (0.05 t CO2 eq/ha yr) for BAU and less than 140 tCO2 eq/yr (0.01 t CO2 eq/ha yr) for the ASC scenario, as given in Figure 4. In both scenarios, the forest fire emissions were very low compared to the total ecosystem carbon stocks (over 200 tCO2 eq/ha yr for the whole period for both scenarios) and stock changes given in Figure 9.

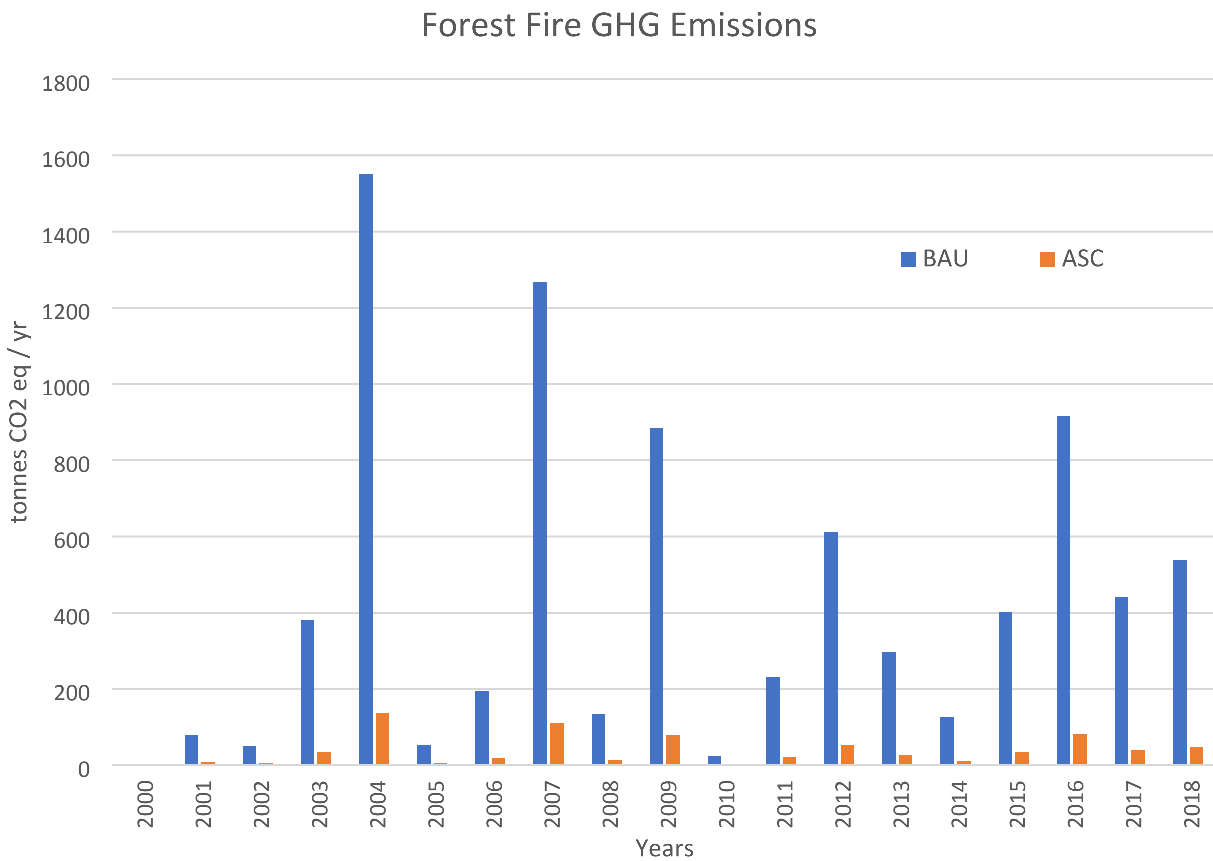


Fig. 4. Total emissions (in tCO_2 eq) due to forest fires for BAU and ASC scenarios for the simulation period.

After 1997's clear-cut event, HWP calculations were based on half-live values. The HWP simulation began in 1997, where the values peaked at around 55 tC. From 1997 onwards, the level remains constant for several years, maintaining its peak value until approximately 2003. Following that period, a gradual decline in tC continued through the years up to 2019 (Figure 5).

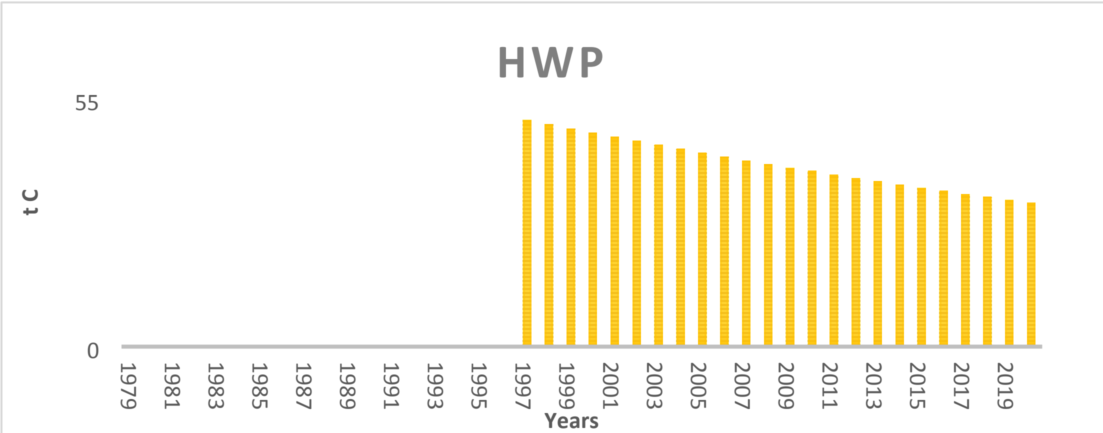


Fig. 5. Harvested Wood Products simulation after the 1997 clear-cut event.

The CBM model was calibrated during the 1990-97 pre-treatment period using the field measurement and existing forest inventory data. The amount of carbon stored in AGB increased during the period as controlled by both growth (Del Río et al. 2017). As shown in Figure 6, pine plantations under the BAU scenario accumulated biomass C faster than broadleaved stands under the ASC scenario. As a result, the BAU scenario stored almost 20 additional tonnes of carbon per hectare by the end of the simulation period.

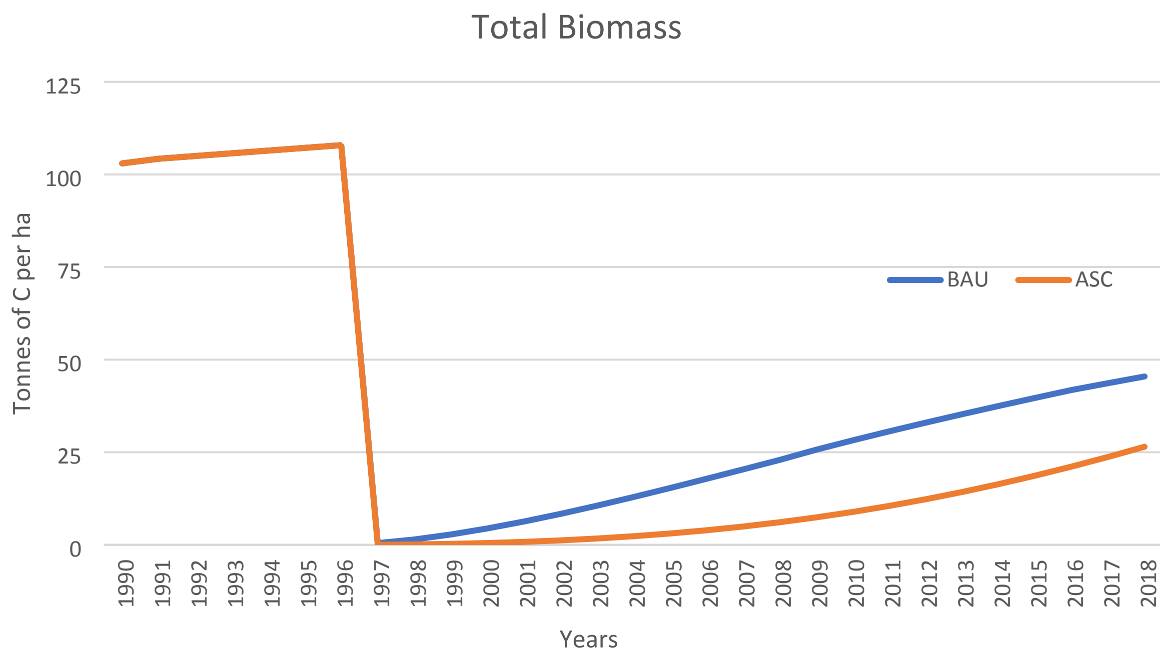


Fig. 6. Change in carbon in biomass for BAU and ASC scenarios from 1997 to 2018.

In 2007, the difference between the scenarios for carbon stocks in living biomass was 15.41 tonnes C/ha. However, in 2018, when the age of the trees was 21, the difference among the scenarios reached a net of 18.97 tonnes C/ha. Despite this, the biomass carbon stocks were still far below the levels before treatment, above 100 tonnes C/ha.

The BAU scenario resulted in a total biomass carbon stock of 45.46 tonnes of C in 2018 due to the increased biomass. On the other hand, the ASC scenario resulted in only 26.49 tonnes of C. The aboveground and below-ground pools followed similar trends as the total biomass.

The conversion accelerated carbon stocking due to an increased growth rate but a higher fire risk since historical records and the literature show that pine forests are much more prone to forest fires than the natural broadleaved stands (Yadegarnejad al., 2015). The aboveground carbon stocks of Maritime pine stands in the BAU scenario were near twice the ASC scenario, reaching 26.48 tonnes C/ha at the end of the simulation period. The below-ground carbon stocks followed a similar trend.

In the first years of both scenarios, field preparation caused a significant mineralization increase, but during the 20 years, the BAU scenario was affected by wildfires greater than ASC scenario.

The litter carbon pool rose over 12 t C/ha following the clear-cuts because of slash and debris leftovers and decreased in the following years (Figure 7). The amount of litter left on the site depends on the stand and the site's ecological properties and logging methods. We expect a fast decomposition following a clearcut that enhances light and moisture dynamics [Roth et al. 2023]. According to the field sampling results, the average litter C stock for Maritime Pine stands was 5.49 tC/ha with a standard deviation of 2.73 tC/ha, while it was 4.18 tC/ha with a standard deviation of 2.89 tC/ha for native broadleaved forests. Therefore, the litter C pool is estimated to increase with age and stabilize around 4-6 tC/ha for both stand types but slightly higher at pine.

The litter carbon stocks decreased for both scenarios (also stand types) for the following two decades with a similar amount rate. However, the BAU scenario litter carbon stocks were slightly higher during the first 5-6 years. The CBM model calculated more litter carbon in the early years for the Maritime pine and then almost equalized due to wildfires and management. The litter C stocks did not replenish during the 20 years and were relatively lower than the pre-treatment period stocks.

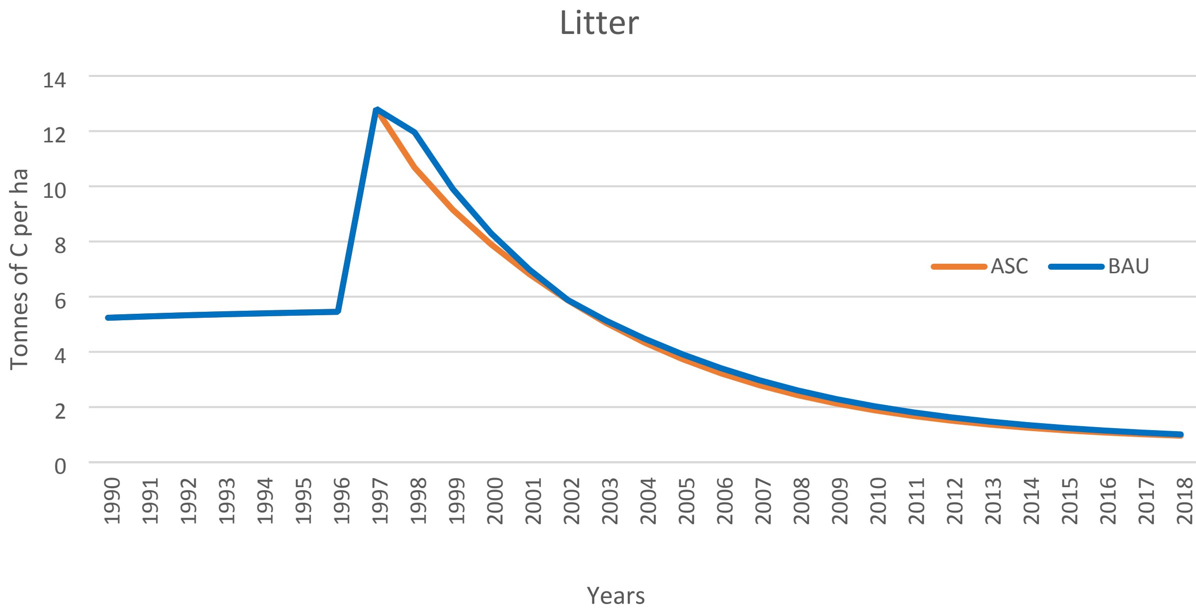


Fig. 7. Litter carbon per hectare for Maritime pine and Mixed Broadleaved stands from 1990 to 2018.

The litter carbon stocks in the BAU scenario were higher than in the ASC scenario for the simulation period. The mineralization in pine stands is generally slower than in broadleaved stands, and a thicker litter layer is typical (Özhan 1977).

Harvest and reforestation affect litter inputs, stand microclimate, and thus soil C storage [Roth et al. 2023]. Also, in our case, soil carbon pools shrank shortly after treatment due to accelerated decomposition caused by field preparation and harvest. The enhanced light and water conditions continued throughout the simulation period of two decades, but for the BAU, the plantations grew fast to establish crown closure and litter accumulation (Figure 8).

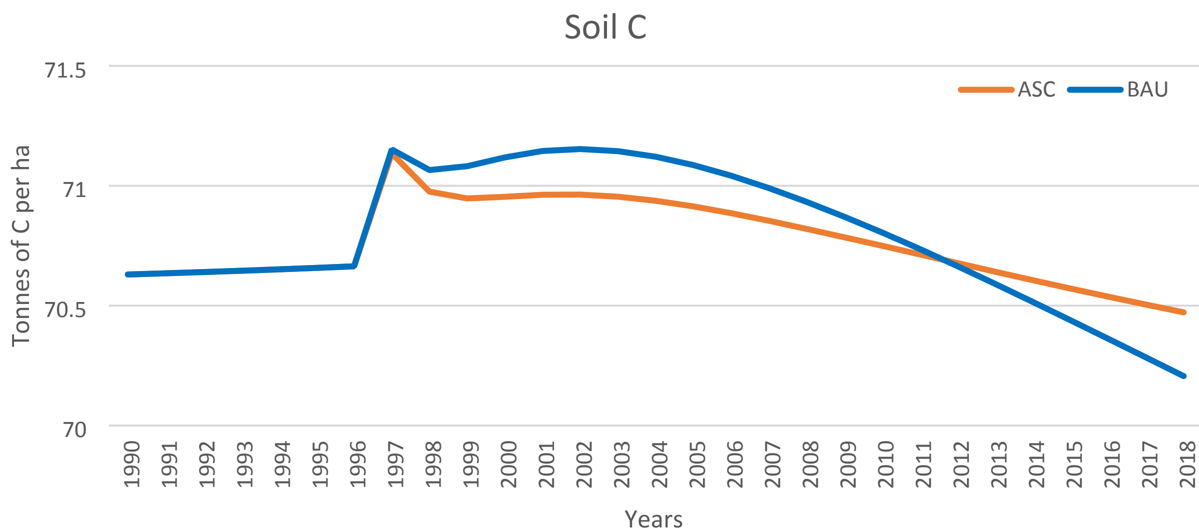


Fig. 8. Soil C per hectare for Maritime pine and Mixed Broadleaved stands from 1990 to 2018.

Slash clearance to reduce wildfire risk might be the reason for reduced soil carbon following harvests. According to James et al. (2021) the effects of slash removal may go on over 20 years after harvest. The authors also underlined the importance of understory vegetation in maintaining soil C, which may not be true for Maritime pine stands. However, the increase in soil C a few years after the plantation can be attributed to the quick growth of the young pine trees.

Around 15 years following the treatments, the difference in soil C between the two scenarios equalized, and the C stocks in the ASC scenario started to get over the BAU scenario. This might be caused by a higher decomposition rate under the Pine plantations. As seen in Figure 7, the litter deposition was higher in Pine stands than in native vegetation, but the decomposition rate was also faster in the soil with more considerable plantation distances and lack of understory vegetation. Early-age silvicultural practices are different among these forest types, which may also affect the C balances of the ecosystems, as underlined by Zhao et al. (2023).

A reason for lower soil C stocks under the Pine stands after 2012 can be attributed to higher fire occurrence rates compared to native forests. However, this should not be a significant driver, considering that only

456 ha of Maritime Pine forest was burnt from 2001 to 2018, around 1.4 percent of the total. Therefore, the decrease in soil C stocks for the ASC scenario during the simulation period can be attributed to decreased accumulation of litter after harvest and partly wildfires. However, the difference in soil carbon stocks between the scenarios was not more than 1 t C per ha.

The ecosystem net GHG balance, which accounts for the difference between removals by growth and emissions by harvest and wildfires, dropped from 650-680 tCO2 eq/ha band to slightly over 300 tCO2 eq/ha in both scenarios due to harvest in 1997 as seen in Figure 9. The carbon stocks rose gradually as the trees aged, and the negative emissions reached 427.8 tCO2 eq/ha and 359.0 tCO2 eq/ha in BAU and ASC scenarios, respectively. The difference in the ecosystem's GHG balances reached up to 68.8 tCO2 eq/ha around 10-15 years following the plantation and remained stable until the simulation period's end. The GHG emission rates remained in ranges of 0.01 - 0.048 tCO2eq/ha and 0 – 0.004 tCO2eq/ha in BAU and ASC scenarios, respectively, during the whole period.

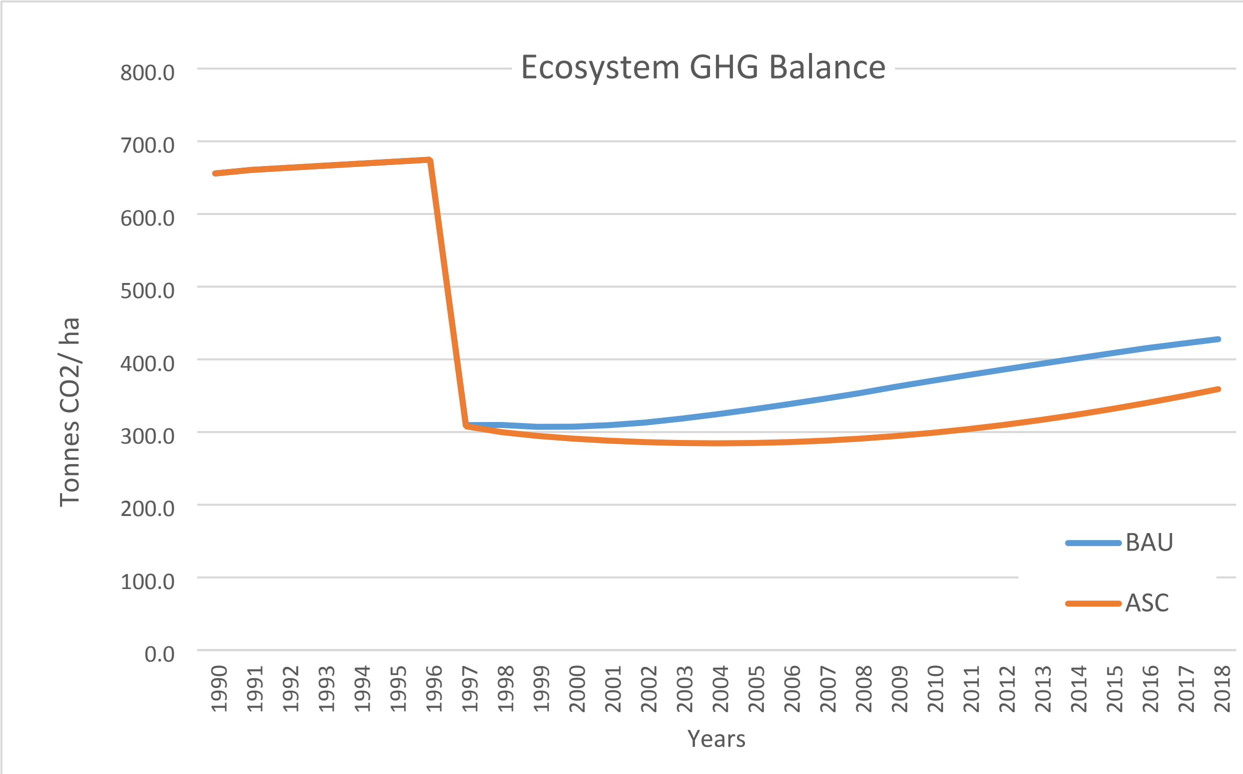


Fig. 9. The Ecosystem Net GHG balance (Removals by growth – Emissions by harvest and wildfires) for the BAU and ASC scenarios.

The difference in AGB carbon stocks between Maritime pine and native broadleaved stands reaching to almost 70 tCO2 eq/hectare in less than 20 years makes the treatment a feasible mitigation option for the Istanbul Region. This is a significant sequestration rate difference to consider fast-growing species as a good mitigation option [Zhao et al. 2023] for ecologically suitable regions with low disturbance risks; however, the rotation period of these plantations is shorter than native forests.

The wildfire affected the pine stands more, but the burnt area was lower than the burnt area of total native forests. Therefore, the wildfire emissions did not reduce the sequestered carbon gains. However, the sequestration rate or disturbance-related emissions are not the only parameters that are evaluated when considering the effectiveness of this mitigation option. The plantations may have other ecological, economic, and social impacts that must be considered. According to Smyth (2023), coniferous plantations may sometimes disturb and drain organic soils, and regulations and safeguards should consider these conditions.

On the other hand, the situation may still change in the coming years, with the increased temperatures causing more dry summer conditions and increased fire risk (Kitzberger et al. 2017; Westerling 2016). Furthermore, urban areas are getting closer to the forests with the city's expansion due to more fires close to urban areas.

Conclusions

Many countries have acknowledged the crucial role of forests in achieving climate neutrality and incorporated mitigation actions aimed at increasing carbon removal into their long-term strategies. However, there have been ongoing debates about the potential risks associated with fast-growing exotic plantations, particularly regarding increased disturbance risks.

Our assessments revealed that in the prosperous plantations, forest growth rates were higher, and total ecosystem C stocks increased faster compared to native broadleaved species. The increased fire emissions of pine stands were ignorable from a GHG balance point of view. In other words, increased fire emissions of converted stands did not significantly harm the enhanced mitigation benefits. However, we should underline that our results are valid in the Istanbul case, where the spread of wildfires was limited due to forest fragmentation and resilient native vegetation. The pine conversions were in patches surrounded by native forests.

Our findings concluded that despite increased wildfire emissions, the fast-growing plantations in Istanbul had more carbon removal benefits. As such, they can be considered a viable option for mitigation in appropriate regions and conditions. However, the adaptation side of the issue, including impacts on wildlife, biodiversity and other disturbances, must be considered to evaluate the whole framework of species conversion.

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Data generated or analyzed during this study are available from the corresponding author upon reasonable request.

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