Ecological Modelling 266 (2013) 144–171

Contents lists available at ScienceDirect

Ecological Modelling

journal homepage: www.elsevier.com/locate/ecolmodel

Application of the CBM-CFS3 model to estimate Italy's forest carbon budget, 1995–2020

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ARTICLE INFO

Article history:
Received 9 November 2012
Received in revised form 5 July 2013
Accepted 7 July 2013
Available online 7 August 2013

Keywords:
Forest modeling
Carbon budget
Greenhouse gas inventory
Uneven-aged forests
Yield tables
National forest inventory

ABSTRACT

The estimation of past and future forest carbon (C) dynamics in European countries is a challenging task due to complex and varying silvicultural systems, including uneven-aged forest management, and incomplete inventory data time series. In this study, we tested the use of the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) in Italy, a country exemplifying most of these challenges. Our objective was to develop estimates of forest carbon budgets of the Forest Management area (including all forests existing in 1990) for the period 1995–2009, and to simulate alternative scenarios of natural disturbance (fire) and harvest rates to 2020. A number of methodological challenges required modifications to the default model implementation. Based on National Forest Inventory (NFI) data, we (i) developed a historic library of yield curves derived from standing volume and age data, reflecting the effect of past silvicultural activities and natural disturbances, and a current library of yield curves derived from the current net annual increment; (ii) reconstructed the age structure for a period antecedent to the reference NFI year (2005), to compare the model results with data from other sources; and (iii) developed a novel approach for the simulation of uneven-aged forests. For the period 2000–2009, the model estimated an average annual sink of −23.7 Mt CO_2 yr^{-1} excluding fires in Italy's managed forests. Adding fires to the simulation reduced the sink to −20.5 Mt CO_2 yr−1. The projected sink (excluding all fires) for the year 2020 was −23.4 Mt CO_2 yr^{-1} assuming average (2000–2009) harvest rates. A 36% increase in harvest rates by 2020 reduced the sink to −17.3 Mt CO_2 yr^{-1}. By comparing the model results with NFI data and other independent studies, we demonstrate the utility of the CBM-CFS3 both for estimating the current forest sink in even-aged and more complex uneven-aged silvicultural systems in Italy, and for exploring the impact of different harvest and natural disturbances scenarios in managed forests. This study demonstrates the utility of the CBM-CFS3 to national-scale estimation of past and future greenhouse gas emissions and provides the foundation for the model's future implementation to other European countries.

1. Introduction

Since the early 1990s, growing concerns about global warming as a consequence of increasing concentrations of atmospheric greenhouse gases have added a new demand for forest ecosystem services. Forests are the second largest carbon (C) stock present in the biosphere, after the oceans (Janssens et al., 2003) and they represent an important C sink that is removing from the atmosphere annually about one third of global fossil fuel emissions (Le Quéré et al., 2009; Pan et al., 2011). The climate mitigation role of forests in industrialized countries has been recognized by the United Nations Framework Convention on Climate Change (UNFCCC) and the Kyoto Protocol (KP) through the emission and removals from the Land Use, Land-Use Change and Forestry (LULUCF) sector. This role has been further confirmed during the recent international climate negotiations (Grassi et al., 2012). In particular, a number of important decisions on LULUCF accounting for the second commitment period of the KP were taken (UNFCCC, 2011), including: (i) the mandatory accounting of forest management, with future emissions and removals being compared against a predetermined "reference level"; (ii) C stock changes in the harvested wood products pool will be accounted; and (iii) emissions and subsequent removals on forest lands affected by natural disturbances may be excluded from the accounting. In most cases, the implementation of these decisions requires the capacity to model the impact of forest management on the current and future C balance of forests, in a way which is consistent with greenhouse (GHG) inventories of the

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0304-3800
http://dx.doi.org/10.1016/j.ecolmodel.2013.07.007

countries; for instance, most countries' reference levels are based on modeled projections of the future C balance under assumed scenarios of business-as-usual harvest (AWG-KP, 2011).

The forest C dynamics can be quantified using (i) empirical models driven by data from national forest inventories (NFI) or (ii) process-based models, driven by the simulation of photosynthesis and other ecological processes. Typically, process-based models have been mainly used to simulate long-term evolution of forest C dynamics (i.e., many decades) including the potential effects of climate change (Pretzsch et al., 2008). While efforts are ongoing to incorporate the impact of forest management into process-based models (e.g., Belassen et al., 2011), empirical models such as EFISCEN (Nabuurs et al., 2000), CO2Fix (Nabuurs et al., 2001) or FORMICA (Böttcher et al., 2008a) still remain the primary tool to simulate the detailed effects of different forest management options in short-term forest C dynamics (i.e., few decades).

Some of the empirical models, such as the CO2Fix V.2, were applied and validated on both even-aged and uneven-aged forests in Europe, Central America and Africa (Masera et al., 2003; Nabuurs et al., 2008) but they cannot directly consider the effects of natural disturbances, such as fires and storms, which may have large impacts on the annual C balance of countries (Lindroth et al., 2009; Stinson et al., 2011). Others, such as EFISCEN (Nabuurs et al., 2000), which have been applied to all the European countries, contain a module to simulate the effect of some natural disturbances (Schelhaas et al., 2002; Seidl et al., 2009) but they generally simplify the silvicultural systems by assuming an even-aged structure for all the forests that is managed by a clear cut system. Although more than 60% of forests are reported as even-aged at the European level, uneven-aged and non-categorized forests cover about 30% and 70% of the total forest area in Central-East and South-West Europe, respectively (UNECE/FAO, 2011a). Moreover, most of empirical forest models can only provide estimates from the reference year of the National Forest Inventory (NFI) onwards. This means that it is not possible to compare empirical model results before the NFI date to the historical data estimated by other sources (e.g., the GHG inventory prepared by the country), and thus a validation opportunity is unavailable.

The current empirical forest models applied to entire European countries have difficulty simulating one or more of the following issues: (i) uneven-aged forests; (ii) natural disturbance events, and (iii) historical estimates of forest C dynamics.

The long-term objective of our work is to quantify past and near future national-scale forest C dynamics of European countries using data from NFIs, including the explicit representation of uneven-aged forest management, the impacts of natural disturbances, and comparing our estimates with historical data from independent sources. Assessing the utility of models to generate reference levels of LULUCF sector emissions and to quantify the outcome of alternative management is of interest to the policy community. We aim to be consistent with the methodological guidance provided by the Intergovernmental Panel on Climate Change (IPCC, 2003, 2006), including the outcome of the recent expert discussion on the use of models (IPCC, 2010). A model needed to meet all these objectives must be sufficiently detailed to accurately represent the flow of C between different pools, and flexible enough to adapt to the complex and varying silvicultural systems, including uneven-aged forests, and ecological conditions typical of most European countries.

Among the available models, the Carbon Budget Model (CBM), developed by the Canadian Forest Service (CFS), appears to meet several of these requirements. The CBM was previously applied at national and regional scales in Canada (Kurz and Apps, 1999; Kurz et al., 2009; Bernier et al., 2010; Stinson et al., 2011) and Russia (Zamolodchikov et al., 2008). It provides the modeling framework and required parameters to simulate natural and human-induced disturbance events (Kull et al., 2006; Kurz et al., 2008; Metsaranta et al., 2010) and the current version of this model (CBM-CFS3, Kurz et al., 2009) meets the IPCC reporting requirements (IPCC, 2003, 2006). However, this model was primarily applied to even-aged forests and has never been applied to an entire European country.

The specific objectives of this study were therefore (i) to test the CBM in different silvicultural systems, proposing a novel approach to include uneven-aged forest structures; (ii) to apply the CBM to a European country, and estimate the forest C balance of the Forest Management area (including all forests existing in 1990, (IPCC, 2003)) from 1995 to 2009 and a projection to 2020, and (iii) to explore the impact on the C balance of different scenario assumptions of future rates of harvest and fire disturbances.

To achieve these objectives, Italy was assumed as a representative case-study of the range of management strategies applied in Europe. This choice was supported by (i) the significant presence of uneven-aged high forests, (ii) a large area of forests affected by fires in the Mediterranean regions, and (iii) the availability of updated data collected through the last NFI in 2005. To be consistent with the definition of forest management under the Kyoto Protocol, in this study we considered only the managed forest area existing in Italy in 1990.

2. Material and methods

This study included methodological developments, and the assessment of different scenarios. Our model assumptions are reported in Sections 2.1, 2.2, 2.3 and 2.5 and the Appendices, and are discussed in the Sections 3.1 and 3.2. Scenarios are defined in Section 2.4 and discussed in Sections 3.3 and 3.4, with detailed comparisons to other studies.

2.1. The Carbon Budget Model (CBM-CFS3)

The Carbon Budget Model is an inventory-based, yield-data driven model that simulates the stand- and landscape-level C dynamics of above- and belowground biomass, and dead organic matter (DOM) including soil (Kurz et al., 2009). The spatial framework conceptually follows Reporting Method 1 (IPCC, 2003) in which, for the purpose of estimation and reporting the spatial units are defined by their geographic boundaries and all forest stands are geographically referenced to a spatial unit. In the present study, the landscape (Italy) was divided into 21 administrative units and 24 climatic units (CLUs, as defined by Pilli, 2012) with mean annual temperatures ranging from −7.5 to +17.5 °C based on climatic data provided by Hijmans et al. (2005). The same approach can be easily extended to all the other European countries. The intersection of the unit boundaries yielded 168 unique spatial units (Fig. 1).

Within a spatial unit, each forest stand is characterized by age, area, and up to 10 classifier types that provide administrative and ecological information, the link to the appropriate yield curves, and parameters defining the silvicultural system (such as forest composition, management strategy and information provided by the Italian National Forest and Carbon Inventory (INFC)).

During the model run, a library of yield tables defines the gross merchantable volume production by age class for each species. These yields represent the volume in the absence of natural disturbances and management practices. The CBM applies the net annual increment (i.e., the periodic increment minus mortality from self thinning) during the model run. Species-specific stand-level equations (Boudewyn et al., 2007) convert merchantable volume production into aboveground biomass, partitioned into merchantable stemwood, other (tops, branches, sub-merchantable size trees) and foliage components. The belowground biomass (coarse and fine roots), its increment and annual turnover are

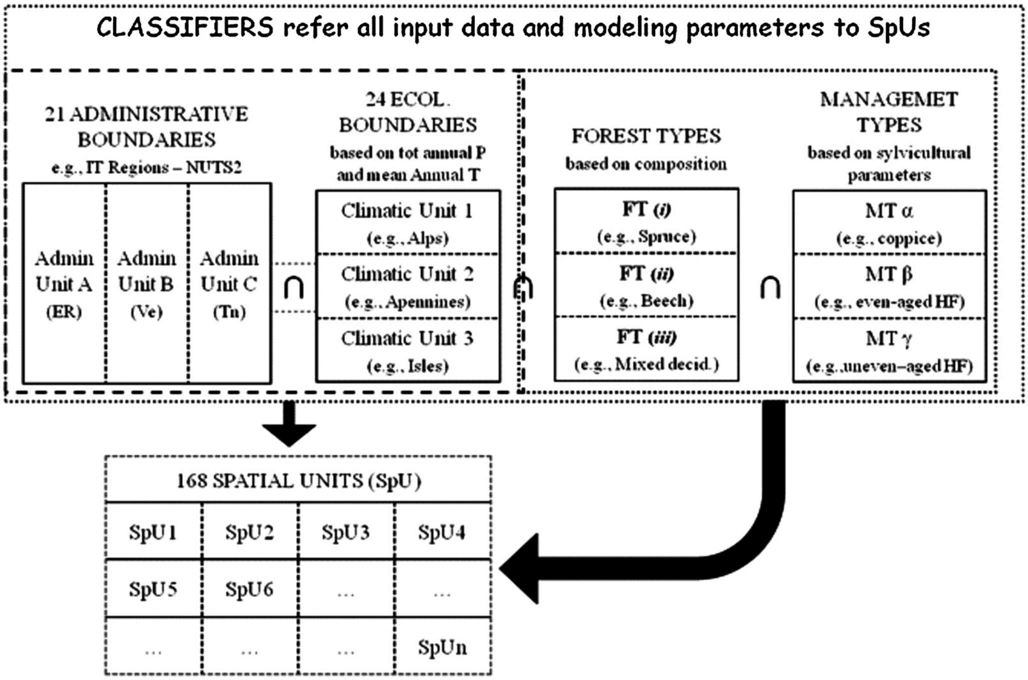

Fig. 1. Schematic representation of the main input data required by CBM in order to define the Spatial Units (SpUs). The dashed line identifies the general classifiers and the dotted line delimits the information provided by the forest inventory, split between each SpU.

calculated using the equations provided by Li et al. (2003). Annual dead wood and foliage input is estimated as a percentage (i.e., turnover rate) applied to the standing biomass stock.

To estimate the decomposition rate of each DOM pool the CBM adjusts the base decomposition rates defined at 10 °C based on the mean annual temperature (T_m) in each spatial unit. Dead organic matter (DOM) pools (dead wood, litter and soil), are initialized using a procedure that takes into consideration site productivity (NPP), temperature-dependent decomposition rates and disturbance history (Kurz et al., 2009). The model starts the initialization process with all DOM pools containing zero C stocks and then simulates multiple iterations of growth and stand-replacing disturbances, gradually increasing the size of the DOM pools. During this preliminary stage, the model can apply the same set of yield tables selected for the main simulation or different historical tables specifically defined for the initialization of DOM pools. The rotations continue until the slowly-decaying C pools at the end of two successive rotations meet a difference tolerance of 0.1%. Once this criterion has been met, the CBM applies a user-selected last disturbance event which affects the amount of C in the DOM pools, and then simulates the stand dynamics to the inventory age of the stand.

In the simulation of stand- and landscape-level carbon dynamics, the user can define natural and anthropogenic disturbances such as fire, insects or storms and partial or clear-cut harvesting (Kull et al., 2006). Users define the amount (area or C target), type and intensity of each disturbance by year and spatial unit (or groups of spatial units). Eligibility criteria, such as forest type, age, or other classifier values can be used to define the eligible stands for each disturbance. Disturbance impacts are defined using a 'disturbance matrix' that describes the proportion of C transferred between pools, transferred to the forest product sector or released to the atmosphere for each disturbance type (Kurz et al., 2009). Post-disturbance dynamics can be specified in considerable detail, allowing for successional pathways and rates of regeneration. Afforestation and deforestation can be also represented as disturbance types with their own disturbance matrices and transitions to and from forest land.

The model provides annual predictions on C stocks and fluxes, such as the annual C transfers between pools, from pools to the atmosphere and to the forest product sector, as well as ecological indicators such as the net primary production (NPP), net ecosystem production (NEP) and net biome production (NBP). The main limitation of the current version of the CBM model is the difficulty in simulating the impacts of environmental changes (e.g. climate) on forest growth because the model does not explicitly simulate the impacts of environmental variations on yields. Annual rates of disturbances are defined by the user and are not calculated as a function of forest conditions or climate, but input data can define future changes in disturbance regimes (Kurz et al., 2008; Metsaranta et al., 2010). In this study, projections were limited to 2020 and the effects of possible environmental changes over this short period have been excluded.

A second constraint of the model is that the CBM input data require forest area by age class and yield tables to quantify the growth rate of each forest type. Such information is generally not available for uneven-aged forests that can represent up to 27% of the area in European countries (UNECE/FAO, 2011a).

The present work used version 1.2 of the CBM-CFS3 model, suitably adapted to the Italian case-study.

2.2. The Italian National Forest and Carbon Inventory (INFC)

Between 2005 and 2008, Italy conducted the measurements and the first implementation step of the new Italian National Forest and Carbon Inventory (INFC). One of the main aims of the INFC was to produce information needed for international reporting such as the FAO (Food and Agricultural Organization of the United Nations), the UNFCCC, and the Kyoto Protocol (Tabacchi et al., 2005). For this reason, the INFC applied the FAO-FRA (Forest Resource Assessment) 2000 forest definition, in order to include categories for both forest and other wooded land use. The INFC identified 17 forest types (FT, reported in Table 1), classified according to the following three categories:

Table 1
Forest types (FT) and area (referred to 2005) identified by the INFC based on the main species list reported in the second column (INFC, 2007a). The last column reports the acronyms adopted in the following tables.

Forest type (FT)	Main species	Area (kha)	Acronym
Oak forests	*Quercus petraea* (Matt.) Liebl., *Quercus robur* L., *Quercus pubescens* Willd.	1084	QR
Oak forests with *Q. cerris*	*Quercus cerris* L., *Quercus frainetto* Ten., other oak species	1010	QC
Mixed deciduous broadleaved for.	Fraxinus ornus L., Robinia pseudacacia L., etc.	994	OB
Beech forests	*Fagus sylvatica* L.	1035	FS
Chestnut forests	*Castanea sativa* Mill.	788	CS
Hornbeam forests	*Carpinus* spp., *Ostrya carpinifolia* Scop.	852	OCa
Norway spruce forests	*Picea abies* (l.) Karsten	586	PA
Holm oak forests	*Quercus ilex* L.	620	QI
Larch and stone pine forests	*Larix decidua* Miller, *Pinus cembra* L.	382	LD
Mediterranean pine forests	*Pinus domestica* L., *Pinus maritima* Miller, *Pinus halepensis* Miller	226	MP
Riparian forests		229	RF
Black pine forests	*Pinus nigra* Arnold, *Pinus laricio* Poir., *Pinus leucodermis* Ant.	236	PN
Cork oak forests	*Quercus suber* L.	168	QS
Scots pine and Mountain pine	*Pinus sylvestris* L., *Pinus uncinata* Mill.	151	PS
Silver Fir forests	*Abies alba* Mill.	68	AA
Other evergreen forests		84	OE
Other coniferous forests		63	OC
"Gaps"	Forest areas that temporarily do not satisfy the parameters (cover, height, etc.) established by the national definition of forest.	53	Distributed between other FTs
Total area		8636	

1. Composition: pure or mixed forests.
2. Forest management type (MT), mainly represented by: (i) even-aged high forests; (ii) uneven-aged high forests; (iii) coppices (with standards, with conifers or singled coppices); (iv) *irregular* (i.e., high forests with different structures on the same area) and *not classified* forests and (v) *special* MT, such as chestnut forests for nut production and coppices (above all beech forests) under conversion to high forests.
3. Growth stage: distinguished between 6 and 7 age classes for even-aged high forests and coppices (Gasparini and Tabacchi, 2011).

2.3. Forest area and climatic parameters

When electing forest management (FM) as voluntary activity under Article 3.4 of the KP for the first commitment period, Italy applied a broad definition of FM (IPCC, 2003; Italy, 2011a), which included all forest area existing in 1990: 7450 kha. This area is considered in this study and corresponds to the forest area reported by the INFC for 2005 (8636 kha, excluding 122 kha of hybrid poplar plantations that according to Italian laws are considered cropland) minus any forest conversion from and to forest that occurred since 1990 according to the Italian National Inventory Report (NIR, Italy, 2011a): on average, about 0.72 kha yr^{-1} of deforestation and 77 kha yr^{-1} of forest expansion for a total net increase of 1186 kha. This study focused on the FM area only, and afforestation was not considered at this stage, even if it represents a relevant aspect of the total carbon balance of Italian forests. In contrast, deforested areas were small (less than 0.01% of the forest management area according to the Italian NIR), and the effect of this disturbance event was also excluded from our analyses.

The total forest area reported by the INFC was first distributed between 21 administrative regions, 17 FTs (as reported in Table 1), 3 MTs (i.e., high forests, coppices and special forest types) and 2 management strategies (MS, i.e., even-aged and uneven-aged forests). For each parameter, some area remained "not classified" in the INFC (on average 9% of the total forest area). Unclassified MTs were entirely assigned to the uneven-aged high forests (which in total included about 3300 kha), while the unclassified area for the other parameters was proportionally distributed between the other classes. Even-aged high forests and coppices (about 5335 kha) were also distinguished between 21 age classes with a 10-year span, starting from the age-class distribution reported by the INFC (Gasparini and Tabacchi, 2011). Even-aged forests not classified for age by INFC, were re-distributed between the other age classes in proportion to the area.

In order to compare our model results with historical data provided at country level for a consistent time period (i.e., 15 years), we reconstructed the past (1995) age-class distribution, starting from the even-aged forests and using the original INFC data for 2005. The age of each stand in the even-aged forests was decreased by 10 years and the corresponding volume reported by the yield tables was applied to the new ages. By applying this rule to stands in the youngest age class (i.e., age <10 years) we obtained some area with a negative age (γ) which indicates that the stand was disturbed and established in the last decade. We assumed that γ was the number of years before the previous stand was affected by a clear cut at age α, established by a set of species-specific silvicultural rules. The resulting age (A) assigned to this stand for the base year (1995) was therefore equal to:

$$A = \alpha \cdot \gamma \tag{1}$$

We then started the simulation in 1995 and applied the clear-cut disturbance to this stand when it reached age α, followed by regeneration, such that the area in the youngest age-class in 2005 approximated the area in the inventory.

This approach allows us to combine the latest available information on forest area, volume and increment to simulate historical emissions and removals for at least 10 years and compare these model predictions to emission and removal estimates included in the country's GHG inventories. A similar approach could be used for other European countries, many of which have recently updated their NFIs. As an alternative approach, we could also use the data provided by the previous Italian NFI, referred to 1985 (MAF-ISAFA, 1988), and validate our model results against the data provided by the last NFI (INFC, 2005). However, the use of the most recent data collected at national level provides a better representation of the recent-past and near-future forest dynamics. Indeed, all the European countries have recently provided estimates on the current (i.e., since 2000) and future (i.e., to 2020) forest carbon dynamics, using the most recent data available at national level (see the Technical Assessment Reports on the Forest Management Reference Level (AWG-KP, 2011)).

For Italy, a comparison between the area reported as even-aged and uneven-aged high forests and coppices by the INFC (2007a) and by the previous Italian NFI (MAF-ISAFA, 1988) highlighted that forest expansion (mainly young natural forests growing on abandoned lands) was primarily assigned by INFC to the not-classified and uneven-aged forest area. Thus, we assumed that the entire area resulting from forest expansion after 1990 was assigned to the uneven-aged MT (which in our study also includes the *not classified* and *irregular* management types, as defined by INFC in Gasparini and Tabacchi, 2011); this MT area was decreased from 3300 kha (data from the INFC) to 2068 kha (the difference of 1232 kha is the assumed cumulative amount of forest expansion from 1990 to 2005).

2.4. General assumptions

2.4.1. Harvest volume

The volume of annual harvest used by the model for the period 1995–2009 was inferred from data provided by the Italian National Institute of Statistics (ISTAT, 2011). These harvest data at national and regional levels also include harvest from plantations (Pilli, 2011). Harvest volumes are largely underestimated because harvest data are mainly based on information collected with different approaches (i.e., the volume of merchantable wood or the amount of area harvested) and provided by different regional authorities (Corona et al., 2007; Chirici et al., 2011). We therefore applied a correction factor (equal on average to 1.5 at the country level) based on the total fellings (i.e., harvest plus residues) reported by INFC at the regional level (Gasparini and Tabacchi, 2011). The correction factor was estimated by comparing, for each region, the INFC fellings for the period 2005–2007 (volume over-bark, considered as felling occurred during the 12 months before the field measurements) with the average data reported by ISTAT for the same period. A 15% reduction for logging residues (IPCC, 2003) was applied to the final felling data (Päivinen et al., 1999; Tabacchi et al., 2010), to obtain harvest values to be used by the model. These values describe the wood volume transferred out of the forest. Overall, for the 1995–2009 period, the average harvest rate used by the model is slightly (4.6%) lower than the one used in the Italian NIR (Italy, 2011a), because in the model we excluded harvest provided from plantations which are not considered as forest under Italian legislation (Pilli, 2011).

For projections to the year 2020 two scenarios of harvest demand were assumed:

Scenario 1 ("*Increased harvest*") projected the 2020 harvest starting from the average of the 2003–2007 harvest data multiplied by a factor of 1.24, i.e., following the assumption of harvest increase applied by Böttcher et al. (2012) and by Italy in the construction of the FM reference level (Italy, 2011b) based on the harvest projected by the PRIMES (for wood for bioenergy) and GLOBIOM (for timber) models. The harvest demand for the period 2010 to 2019 was linearly interpolated using 2009 and 2020 data. The final amount of harvest predicted for 2020 is about 36% higher than the average historical harvest reported between 2000 and 2009 (i.e., 30% higher than the 2009 harvest demand). This scenario was consistent with the assumptions recently proposed by Italy, and other European countries, in the submission of information on forest management reference level (Italy, 2011b). A sensitivity analysis for this scenario was also carried out, based on a ±10% variation on the total harvest for the period 2010 to 2020.

Scenario 2 ("*Constant harvest*") assumes that the harvest rate for the period 2010–2020 was equal to the average harvest rate observed from 2000 to 2009.

The harvest demand assumed for 2020 was 11.39 and 15.49 million $m^3\,yr^{-1}$, respectively for the constant and increasing harvest scenarios (Fig. 2). We assumed that this demand was totally satisfied by the FM area, and excluded any allocation of harvest to the forest expansion after 1990 (not considered in our study). To distribute the total harvest demand between different regions, forest types and silvicultural treatments, we used the assumptions reported in Appendix A.

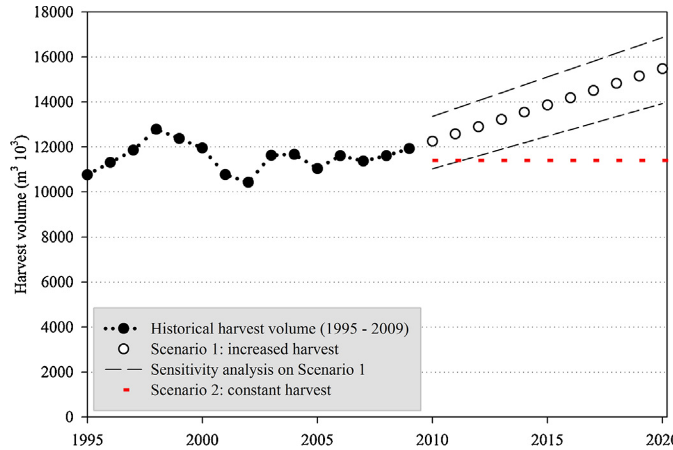

Fig. 2. Italy's harvest volume (excluding logging residues) based on historical (1995–2009) and projected (2010–2020) data assuming for 2020 a 36% increase on the average historical harvest demand for the period 2000 and 2009 (Scenario 1, Increased-harvest) or a constant harvest demand (Scenario 2, Constant-harvest-). The dashed lines from 2011 refer to the values applied for sensitivity analysis (i.e., ±10% variation of the average harvest rate).

2.4.2. Fire data and scenarios

The total forest area affected by fires was derived from the Italian NIR (Italy, 2011a). This figure was adjusted to exclude the area of fires occurring in the area of forest expansion after 1990 (not included in this study). In order to allocate the 1995 to 2010 time series at the regional level, the area reported by NIR was divided according to the proportion of burned forest area reported, for each region and year, by the Italian State Forestry Service (CFS web site) and by the European Forest Fire Information System (Schmuck et al., 2011). If data were unavailable at the regional level (i.e., for 1995, 2004, 2005 and 2006) we filled these data gaps using the average regional distribution of area burned for the other years (1995–2010). The distribution of fires between FTs at the regional level was derived by data reported by INFC (INFC, 2007b).

Assuming that the probability of being affected by fire was also proportional to the mean annual temperature and inversely related to total annual precipitation, the total forest area burned for each year, region and FT was split between each CLU. For each year and region, the whole burned forest area was first assigned to the warmest CLU and only if the burned area was more than the warmest CLU's area, to the other CLUs (according to a decreasing gradient for the mean annual temperature and an increasing gradient for the total annual precipitation). Within each CLU, FT and region, fires were distributed between each MT according to its proportion in the total forest area. The same approach could be applied to any other European country where no detailed information on the distribution of fires (among species and climatic units) is available.

The default disturbance matrix for fires provided by CBM was modified to assume that in the area affected by wildfire, 30% of aboveground biomass was not disturbed (Tabacchi et al., 2010; Pettenella and Ciccarese, 2009). The remaining amount of biomass was assumed killed and moved to the DOM pool (mainly from the merchantable pool) or consumed by fire (about 20% of the leaves and of the other biomass pools). Fires also consumed some of the DOM pools. All organic matter consumed by the fire was released to the atmosphere as CO_2, CO and CH_4.

We developed the following simulation scenarios for fires: (a) no area affected by fire for the entire period 1995–2020; (b) historical fire data for the period 1995–2010, and minimum level of

Table 2
Main parameters characterizing the model scenarios applied to the Italian case study.

Scenarios	Description	Sensitivity analysis	Fires
1a: Increased-harvest-no-fire	Historical harvest rate + increasing harvest demand predicted by economic models (Fig. 2)	±10% variation on the 2010–2020 harvest rate	No
2a: Constant-harvest-no-fire	Historical harvest rate + constant average harvest demand applied since 2010 (Fig. 2)	No	No
2b: Constant-harvest-Min-fire	Scenario 2 + historical fires disturbances + minimum level of wildfire, since 2010	No	Yes (Fig. 11)
2c: Constant-harvest-Avg-fire	Scenario 2 + historical fires disturbances + average 1995–2009 level of wildfire, since 2010	No	Yes (Fig. 11)

historical fire area (detected in 2006) for the period 2011–2020, and (c) historical fire data for the period 1995–2010, and average level of historical fire data for the period 2011–2020.

These fire scenarios were combined with the harvest scenarios to generate the following four simulation scenarios (Table 2): (1a) Increased harvest-no-fire; (2a) Constant harvest-no-fire; (2b) Constant harvest-Min-fire; (2c) Constant harvest-Avg-fire.

Fig. 10 contains the time series of area burned annually for the period 1995–2010 and from 2011 to 2020, assuming the minimum (i.e., Scenario 2b, with about 14,100 ha yr^{-1}) or the average (i.e., Scenario 2c, with about 36,400 ha yr^{-1}) of the area burned in the previous period.

2.5. Growth models

Italy contains a wide range of forest species managed by different silvicultural systems due to the geographic position and morphological characteristics of the peninsula. Our approach was to develop growth curve libraries for both historic and current even-aged forests, and to develop a novel method to estimate growth in uneven-aged forests. We also defined a set of standard silvicultural treatments, that specified percentage reductions in volume (and therefore biomass C) and transfers from the merchantable pool to the forest product sector.

2.5.1. Even-aged forests

The CBM model requires yield tables (YT) representing the gross merchantable wood volume (including decay, waste and breakage anticipated during the logging operation and excluding self-thinning, Kurz et al., 2009). To select a set of YTs suitable for the model, we started our analysis from the current annual increment (CAI) and the average standing volume reported by the INFC for each FT and region.

Due to a long silvicultural tradition, a large number of YTs are available for European countries, representing the development of the main stand-level forest parameters for defined treatments and yield classes (Pretzsch et al., 2008). The values of standing volume per hectare and age class that were used to construct these YTs, and similarly the average volumes per hectare reported by the NFIs, represent the volume of trees remaining after natural disturbances and silvicultural practices (e.g. thinnings). Therefore, the YTs typically available in the literature (or derived from average volumes per ha reported by the NFIs) describe the historical evolution of the volume resulting from past human (i.e., silvicultural practices) and natural disturbances. By contrast, the CAI reported by the INFC, such as by many other European NFIs, represents the *net annual increment*, i.e., the average annual volume of gross increment less that of natural losses (i.e., self-thinning) of all living trees (TBFRA, 2000). The volume detected in a stand at a given point in time is the sum of the CAIs up until that point, minus the sum of losses from natural disturbances and silvicultural activities.

This important difference, shown in Fig. 3, has been described by Pretzsch (2009) as the intermediate volume yield factor (IY_v):

$$IY_V = \left(1 - \frac{S_V}{GY_V}\right) \times 100 \quad (2)$$

where S_V is the standing volume (linked to the NFI volume) and GY_V is the gross volume yield (linked to the NFI CAI). Depending on species, age, stand conditions and silvicultural treatments, IY_V can vary between 33% and 80% with higher values in mixed stands (Pretzsch, 2009).

In our case, the YTs derived from the average volume reported by the INFC may adequately represent the standing volume and the current aboveground biomass stock but would underestimate net increment. YTs representing net increment can be derived from the CAI values per age class reported in the INFC. These tables, however, would overestimate the current aboveground biomass stock because they do not account for losses from natural disturbance and intermediate harvest removals (Fig. 3).

Given these relationships, two different sets of YTs were needed to estimate the initial standing volume and to run the model. The first set, called the 'historical library', was derived from the standing volumes per age class reported by the INFC. This library was used only: (i) in the simulation-initialization procedure (see Section 2.1), and (ii) to estimate the standing volume at the start of the simulation (1995), obtained by assigning the volume of each class (by FT and region) to the 1995 age classes distribution (see Section 2.3).

The second set of tables, named 'current library', was derived from the CAI values reported in the INFC and was applied during the model runs (from 1995 onward) to estimate the current *volume increment* of each stand. During the model simulation, the volume increment predicted from the current library will be reduced by disturbances and silvicultural practices.

Fig. 3 describes the steps followed in our work: set up of the historical library based on INFC standing volume data, estimation of time-zero (i.e., 1995) standing volumes, and model runs using the current library YTs based on INFC CAI values.

The historical YTs library was developed from a large species-independent database, including about 1460 equations derived from the European forest yield tables database (AFOLU database, Teobaldelli et al., 2007) and from an Italian literature review (Castellani, 1982). Because these tables were based on direct field measurements collected by forests subject to management practices for a long time (such as most European forests), they adequately represent the historical evolution of the standing volume on sites directly affected by human activity. All the original data provided by the YTs were interpolated through a Chapman-Richard function (Richards, 1959). Parameters were estimated using the Marquardt method (Motulsky and Ransnas, 1987) provided by the SAS® software, to estimate the merchantable volume for 21 age classes between 10 and 210 years. We created a species-independent database of general equations (that we named UBALD)

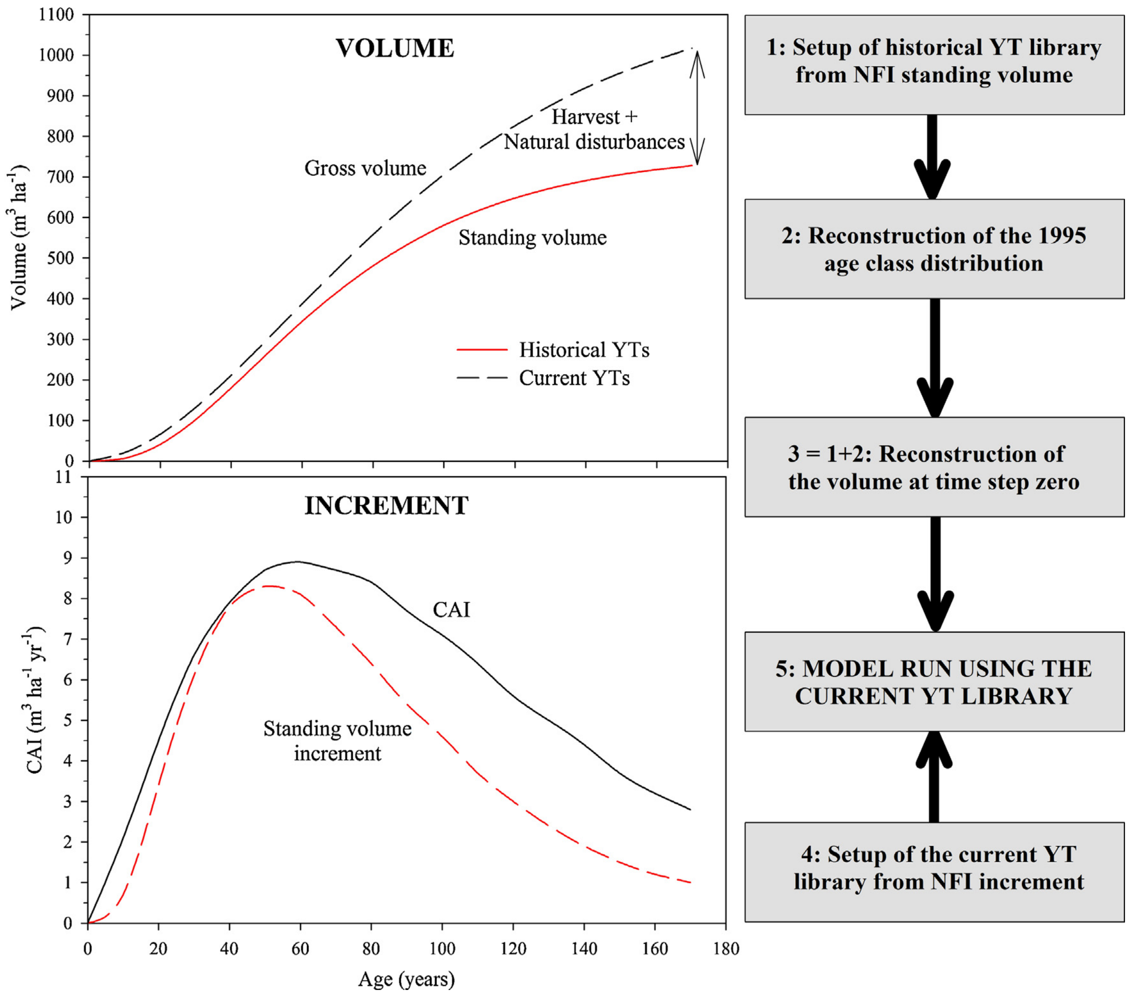

Fig. 3. The upper and lower panels report an example based on the yield tables applied for beech high forests, of the volume and the corresponding increment values applied by CBM. The main steps developed by our approach are indicated on the right. The historical YT library derived from NFI standing volume (step 1, red solid line in the upper panel) is applied to the 1995 age-class distribution (step 2) to estimate the standing volume at the beginning of the simulation (step 3). The current YT library (black dashed line in the upper panel) is derived by the NFI current annual increment (CAI, step 4, black solid line in the lower panel) and applied during the model run (step 5). The difference between the current and historical volume (i.e., about 31% at 180 years) is due to natural disturbances and past management practices. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

and calculated for each equation the average volume. We selected the equation having the minimum relative difference from the average volume reported by the INFC for each FT (distinguished between even-aged high forests and coppices) and region. These equations were then used to compile the historical library.

The current YTs library was derived from the original CAI values reported by the INFC (for each age class, region and FT), corrected to account for the amount of young trees that exceed the minimum diameter at breast height (*dbh*) threshold, during one year (Tomter et al., 2012). According to data inferred by Tabacchi et al. (2010), this amount was equal (at national level, considering all the FTs together) to about 10% of the total CAI reported by the INFC. For each FT and region, we estimated the CAI_t evolution against time using the following combined exponential and power function (Sit, 1994):

$$CAI_t = at^b c^t \tag{3}$$

where t is the average age reported by the INFC for each age class, the parameter a controls the maximum increment reached by CAI and parameters b and c (assuming for our study $b > 0$ and $0 \leq c \leq 1$, according to the values proposed by Sit, 1994) control the shape of the curve. Parameters were estimated using the Marquardt method (Motulsky and Ransnas, 1987) provided by the SAS® software. The YTs applied to the current library were directly derived by the values of CAI_t estimated by Eq. (3), avoiding the use of any empirical table. The theoretic evolution of the CAI for even-aged high forests and coppices, respectively, based on the YTs derived by Eq. (3) is reported in Appendix B.

The approach described above could be applied to any other European country because it is based on assumptions about YTs which can be generalized to all European countries, and because it uses volume and increment data directly derived from available NFIs.

2.5.2. *Uneven-aged high forests*

The silvicultural system applied to uneven-aged forests is not based on an age-class distribution but on the measurement of stand density and the desired diameter distribution (Bettinger et al., 2009). The key-parameters generally considered for these stands are *dbh* and height (Gul et al., 2005). Therefore, YTs, which assume that all stands are fully stocked, pure and even-aged, cannot be directly applied (Trasobares et al., 2004).

An uneven-aged structure is an artificial system (Ciancio et al., 2006) which is dependent on the continuous application of the silvicultural treatment, generally a cutting cycle of 12–20 years (Schutz, 1997; O'Hara, 2001). We therefore assume that the majority of forests reported as uneven-aged in Italy (as in other parts of Europe) are currently (or were, in the recent past) managed through a single tree or a group selection method. According to INFC, (i) at least 34% of the total forest area in Italy was not recently affected by silvicultural practices and (ii) only 1300 kha of forests are currently managed through a partial cut system, related to an uneven-aged or irregular forest structure. These stands (which we assume include 950 kha of not-classified management types) would probably evolve, over a long period of time, towards an older even-aged structure (Cappelli, 1991; Del Favero, 2004). However, due to the short time period covered by our analysis, we did not consider this ongoing transition in silvicultural practice.

The lack of suitable data and the inability to directly apply variables such as age and site index, make the modeling of growth in uneven-aged stands difficult (Peng, 2000) and prohibit the application of empirical, mechanistic or hybrid models, or of age-independent equations (Tomé et al., 2006). To overcome these gaps, and to apply a yield-driven model to these forests, we considered the following aspects characterizing uneven-aged stands (Colpi and De Mas, 1992):

(a) The age is strictly related to the silvicultural treatment applied to the stand: a selective removal of single trees or groups of trees (i.e., a partial cut) in the dominant crown class in order to favor the lower crown classes.
(b) The merchantable volume of the first age class is not null, but is equal to the volume of trees left after the partial cut, when 15–20% of the volume is removed.
(c) Natural mortality is quite limited because of the periodic removal of trees through partial cutting.
(d) The removal of trees has a positive effect on the overall CAI of the stand because partial cutting mainly removes the oldest trees that generally have a lower CAI. In fact, the CAI will increase immediately after the cut in absolute terms or as a proportion of the remaining biomass, and it will decrease progressively during the following years (Hellrigl, 1973; Bettinger et al., 2009).
(e) In Italy, as highlighted by the results provided by the last NFI (Gasparini and Tabacchi, 2011), the current silvicultural practices remove less volume than is added through CAI thus leading to an accumulation of biomass.

Starting from these considerations, and taking into account the specific information reported by the INFC (Tabacchi et al., 2005; INFC, 2009), the silvicultural model proposed for uneven-aged stands was based on the following assumptions:

(a) The average volume (V_0) reported by the INFC for each uneven-aged and irregular high forest type was assumed as the reference merchantable volume and assigned to the reference age class *X*. V_0 is the volume of trees left after the partial cut. For the purpose of running the model, age class *X* was arbitrarily assigned to age class 3 (i.e., 20–30 yrs), which we assume here to represent the average volume of an uneven-aged forest (see Fig. 4, panel B). This approach was implemented for each FT and region.
(b) At the end of the cutting cycle (between 12 and 25 years, according to the literature), a partial cut was applied to the uneven-aged stands, assuming a 15–20% reduction in biomass C (depending on species) and a transfer of the merchantable C to the forest product sector.
(c) The volume (V_t) of the following years (from age class 4) was estimated as:

$$V_t = V_{t-1} \times Ip \tag{4}$$

where V_{t-1} is the volume of the previous year (for $t = 1$, V_0) and *Ip* is the percentage increment estimated as:

$$Ip = \frac{CAI}{V_0} \times 100 \tag{5}$$

where *CAI* is the current annual increment ($\mathrm{m}^3\,\mathrm{ha}^{-1}\,\mathrm{yr}^{-1}$) reported by the INFC for each uneven-aged and irregular forest type and region and V_0 is the average volume reported for the same category ($\mathrm{m}^3\,\mathrm{ha}^{-1}$). For the reference class (age class *X*) the increment was assumed equal to the *Ip* value estimated by Eq. (5); for the subsequent classes it was estimated through the following exponential function (Sit, 1994):

$$Ip_t = ab^t \tag{6}$$

where Ip_t is the percentage increment estimated for the year *t*, *a* is the maximum value on the *y* axis (i.e., the *Ip* value estimated by Eq. (5)) and parameter *b* (with $0 < b < 1$) controls the rate at which the curve approaches its asymptote on the *x* axis. Based on a preliminary sensitivity analysis a constant *b* value of 0.98 was assumed (see Appendix C for further details).
(d) When *Ip* was reduced to zero, the volume was maintained constant.
(e) The volume assigned to age classes 1 and 2 was equal to the volume assigned to age class 3, reduced by the same percentage increment assigned to the reference age class.

Based on these assumptions and on the data reported by the INFC, species-specific yield tables were developed for the uneven-aged forests of each region. The tables, reporting the aboveground biomass ($\mathrm{m}^3\,\mathrm{ha}^{-1}$) of the main representative FTs, were divided into 21 age classes with a span of 10 years, assuming a volume equal to 0 for the first age class (age class 0) and V_0 for age class 3, at 30 years.

Since these forests are not classified according to age, the area reported as uneven-aged was entirely assigned to the reference age class *X*. After the partial cut, the area affected by this disturbance is always transferred back into the reference age class *X*, through the parameter '*age following the disturbance event*' which is controlled by the CBM user (Kull et al., 2011).

All the theoretical assumptions described in this paragraph can be directly applied, at least at the European level, to any other uneven-aged forest or to other forests not explicitly classified as even-aged (i.e., about 27% of the total European forest area), provided that relevant data are available for these forests.

We identified three critical parameters required to simulate uneven-aged stands:

(a) The average volume (V_0) reported by the INFC: this value affected the initial volume assigned to age class 3 and the volumes for higher age classes through the percentage increment.
(b) The CAI reported by the INFC: it affected the volume assigned to age classes 4 and the overall growth rate.
(c) The frequency and intensity of single tree or group selection system: in this study we assumed that each partial cutting event removed on average between 15% and 20% of the merchantable biomass. This was based on published values and from economic and silvicultural reasoning (Del Favero, 2000). A rate below 10% would not be cost effective, and a utilization rate above 20% at a cutting cycle of 15–20 years would not be sustainable for an uneven-aged silvicultural system.

We explored the impact of these parameters on final results using a test dataset and determined: (i) the effect of a 20% variation of V_0 and *CAI* on the aboveground C stock, through a sensitivity analysis performed on the data provided for spruce in the Trentino region; (ii) the effect produced by different cutting cycles, through

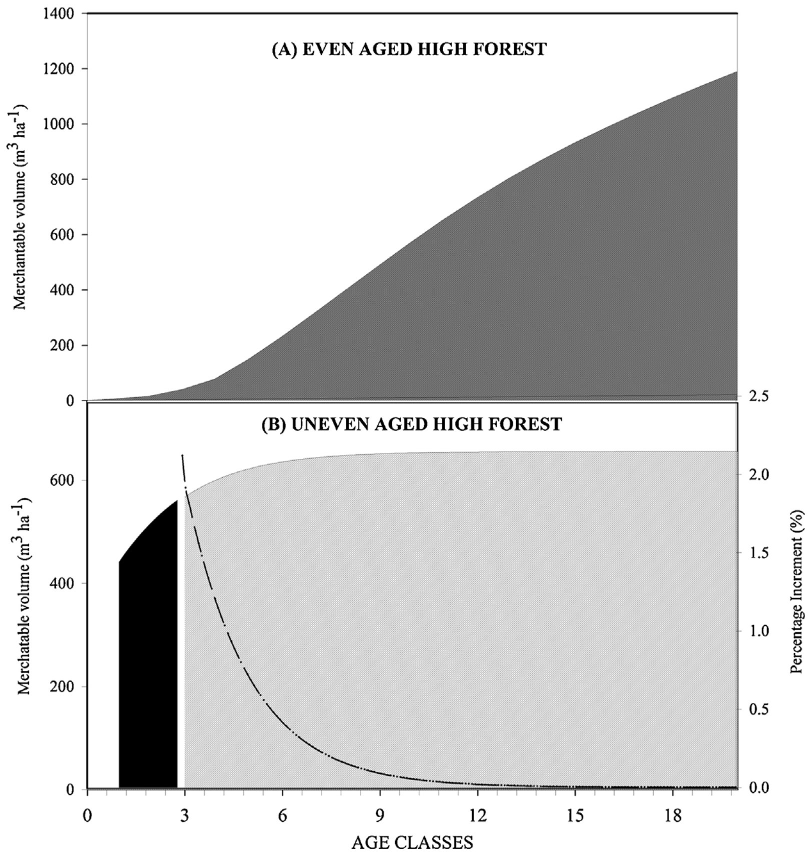

Fig. 4. (a) Even aged high forest: yield curve for spruce even-aged high forest for a spruce forest, for Trentino region; (b) Uneven aged high forest: merchantable volume (grey area) applied to a spruce uneven-aged high forest for Trentino region, developed applying a decreasing percentage increment (dashed black line) to the average volume reported by the INFC for the spruce forest type of the same region. The white line represents the reference age class (3); volume assigned to age classes 1 and 2 was equal to the volume assigned to age class 3, reduced by the same percentage increment assigned to the reference age class (black area).

a variation of 10 years compared to the theoretical cycle assumed for 4 representative species for the same region.

2.6. Calibration of the stand-level equations and DOM parameters

To estimate the aboveground biomass from the volume-based yield tables, the CBM applies species-specific stand-level equations developed by Boudewyn et al. (2007) for each ecozone and province, as defined for Canada. Each of the Italian forest types were associated to an appropriate Canadian species following the approach described in Appendix D.

The CBM simulates dynamics of dead organic matter and soil C using a process-based approach (Kurz et al., 2009). The model uses biomass turnover rates and litterfall transfer rates to represent annual biomass mortality (including trees, leaves, branches and roots) and biomass C transfers to DOM pools. Fig. 5 summarizes the fluxes of C between the main pools. During the simulation, merchantable C moves to the snag stem pool (part of the dead wood pool), foliage moves to the very fast aboveground pool and other wood compartments are moved to the snag branches (part of the dead wood pool) and the aboveground fast pool (part of the litter pool). Snag stems transfer C to the medium aboveground pool and snag branches transfer C to the fast aboveground pool.

Dead coarse and fine roots move to the aboveground and belowground fast and very fast DOM pools. The biomass turnover rate (e.g., % mortality yr^{-1}) is defined for each live biomass pool (Kurz et al., 2009). The decomposition of DOM pools is modeled using a temperature-dependent decay rate that determines the amount of organic matter that decomposes each year in each DOM pool. The decayed C is released to the atmosphere or transferred to the more stable slow DOM pools. Further details are provided in Kurz et al. (2009) and Smyth et al. (2010).

To calibrate the DOM parameters, we adjusted turnover, decay and spin-up parameters by comparing model predictions to data reported in the literature for some Italian regions, (see Appendix E). The same method could be applied in other European countries.

The final values of the pool-specific base decay rates at the reference temperature (10 °C) that were used for the CBM application in Italy are reported in Fig. 5.

3. Results and discussions

The following sections present the main results of the study and discuss: (i) the main methodological assumptions related to implementation of CBM for even-aged (Section 3.1) and uneven-aged (3.2) forests; (ii) the dynamics of C stock changes as estimated by the model (Section 3.3); and (iii) our results in comparison with other studies and with Italian NFI data (Section 3.4).

3.1. Even-aged growth model

Two main challenges that were addressed by our study for the even-aged forests were: (i) the different meaning of the volume and increment data reported by the NFI and (ii) the correct reconstruction of the historic age-class distribution.

In stands managed over a long period of time, such as many European forests, the current standing volume in inventories is mainly affected by past silvicultural practices. In contrast, the

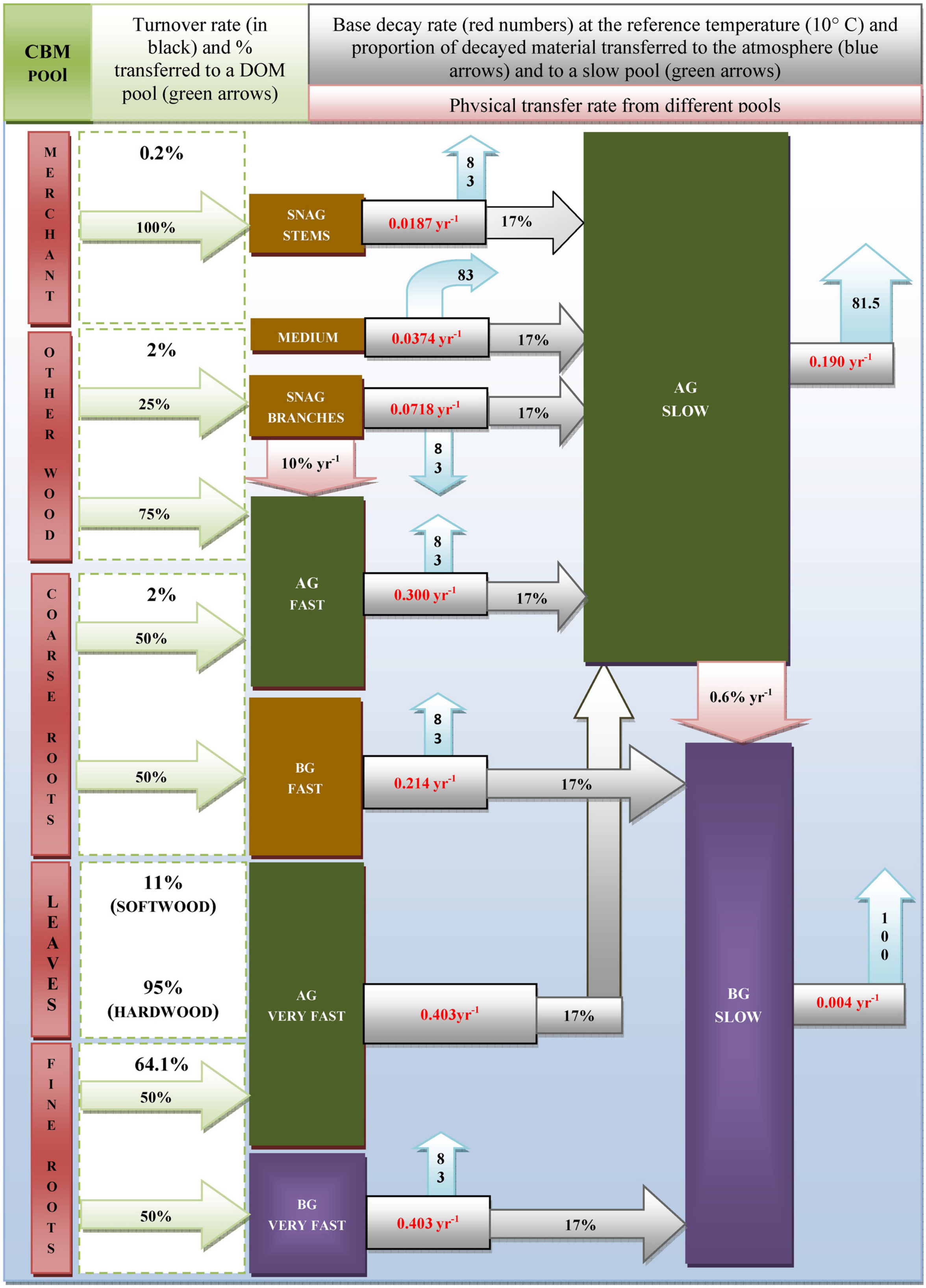

Fig. 5. Biomass turnover rates and DOM dynamics parameters applied for model simulation (see Table 2 reported by Kurz et al., 2009 for a detailed description of each pool). The figure reports the base decay rates (red numbers) at the reference temperature (10 °C). The actual rates vary across the country according to mean annual temperature defined by CLUs. AG = aboveground pool BG = belowground pool. Colors represent the correspondence between CBM and GPG DOM pools: brown for dead wood, green for litter and purple for soil. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

volume increment in inventories reports the current growth of stands, as estimated from direct field measurements, and can be influenced by changes in stand fertility and climatic conditions. This difference, highlighted by Fig. 3, is of limited concern in countries without thinning interventions prior to final harvest, but it is very important for most European countries, where yield tables (YTs) are mainly derived for forests that have been subject to management practices for a long period of time.

As described above (Section 2.5.1) we used two YT libraries in our study. The historic YTs were used for the model initialization and the current YTs were used for the simulation over the period 1995–2020. The average annual increment estimated by the historic and current YTs libraries was equal to 2.7 m^3 ha^{-1} yr^{-1} and 5.1 m^3 ha^{-1} yr^{-1}, respectively. This last figure is consistent with the average CAI reported by the INFC for the even-aged forests, equal to 4.7 m^3 ha^{-1} yr^{-1} (a detailed comparison between these data is reported in Figs. B1 and B2, in Appendix B). The adequacy of our approach is confirmed by the good match between the aboveground biomass stock and the net growth estimated by the model and the one derived from INFC (see Table 5 and the following section).

Both YT libraries were based on species-independent YTs, without the need to collect local YTs. This approach substitutes the growth functions applied by other yield data driven models such as EFISCEN (Schelhaas et al., 2007) or FORMICA (Böttcher et al., 2008a) and it reflects the different theoretical meaning of the volume and increment data provided by many European NFIs. This also overcomes the limitations suggested by other authors (i.e., Röhle, 1995) resulting from the use of old YTs (as in the case of Italy) to estimate current growth and carbon stock of forests. Yield models based on past field measurements may not adequately represent current forest growth due to the influence on growth of increased air temperature, anthropogenic nitrogen deposition, elevated atmospheric CO_2 concentrations, and changes in the forest management activities (Mund et al., 2002; Hember et al., 2012).

For the even-aged forest, we reconstructed the 1995 age-class distribution assuming a constant rotation length for each FT, similarly to what was reported by other authors (Böttcher et al., 2008a; Bellassen et al., 2011). The ability of our approach to duplicate the age-class distribution reported by the INFC for 2005 (Gasparini and Tabacchi, 2011) is confirmed by comparing in Fig. 6 the values 2005-CBM with 2005-INFC. This figure also shows the changes in the age-class structure from 1995 to 2020 for both even-aged high forests and coppices. Since during this period, about 28,360 ha yr^{-1} of coppices older than 60 years were converted to high forests (see Appendix A for further details), the total coppice area reported by the figure decreased during the simulation run.

We further assumed that only the forest area reported for 2005 in the youngest age class could have been affected by clear cut during the previous 10 years. In accordance with this assumption, we inferred that clear cuts supplied, on average (between 1995 and 2009), 9% of the total harvest demand. This figure is consistent with the INFC field measurements which detected the use of a clear cut system on about 6% of the total forest area affected by silvicultural practices (Gasparini and Tabacchi, 2011). Indeed, the current Italian laws strongly limit the use of clear cuts, especially in high forests.

According to our results, the remaining harvest demand was supplied by thinnings in even-aged forests (62%) and partial cuts in uneven-aged high forests. These forests (also including, in our study, the area not classified for the management types), cover about 30% of the total forest area and provided 29% of the total harvest demand between 1995 and 2009. The same thinning share used by this study was applied to Italy by the EFISCEN model in the European Forest Sector Outlook Study II (EFSOS II study) for the scenario aimed at maximizing the biomass C stock (UNECE/FAO, 2011b).

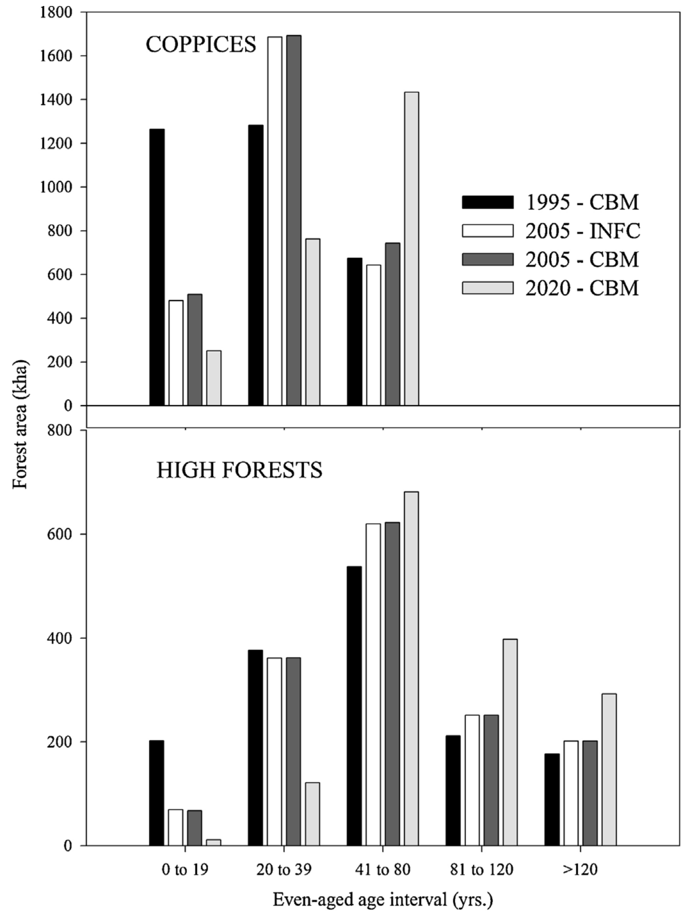

Fig. 6. Age-class evolution of even-aged high forests (excluding coppices converted to high forests during the simulation) and coppices (only considering stands younger than 60 years) between 1995 and 2020, based on the historical harvest and fire rates estimated until 2009 and the projections for the scenario Constant-harvest-Min-fire. The figures reported as INFC–2005 were inferred by Gasparini and Tabacchi (2011).

The two main objectives of our approach were to (i) correctly reconstruct the reference NFI age structure (i.e., 2005) for even-aged forests and (ii) to satisfy the total harvest demand. This approach is similar to the methods proposed by Kurz and Apps (1999) and Bellassen et al. (2011), but unlike Bellassen et al. (2011) we assumed a constant rotation length in even-aged forests. This choice was related to the quite limited area affected by clear cuts and the relatively short time period considered in our study. Moreover, in coppice stands (providing about 90% of the total clear cut amount) the final rotation length is limited to between 20 and 40 years as many species have a decreasing capacity to regenerate from sprouts or root suckers at higher ages.

3.2. Uneven-aged growth model

A novel approach has been developed to implement CBM to uneven-aged forests. Essentially, for each forest type (FT) all the uneven-aged forest area was allocated to a reference age class having the average volume reported by the NFI. Starting from this age class, a decreasing percentage increment was applied to the following age classes (see Eq. (6)). This approach was tested through a number of simulations varying different parameters.

Fig. 7 shows the results of a 20% variation in the average CAI and volume of spruce, which indicates a positive correlation between the aboveground C stock and the CAI and a negative correlation between the aboveground C stock and V_0 (i.e., the average volume reported by INFC). According to Eq. (5), an increase in V_0

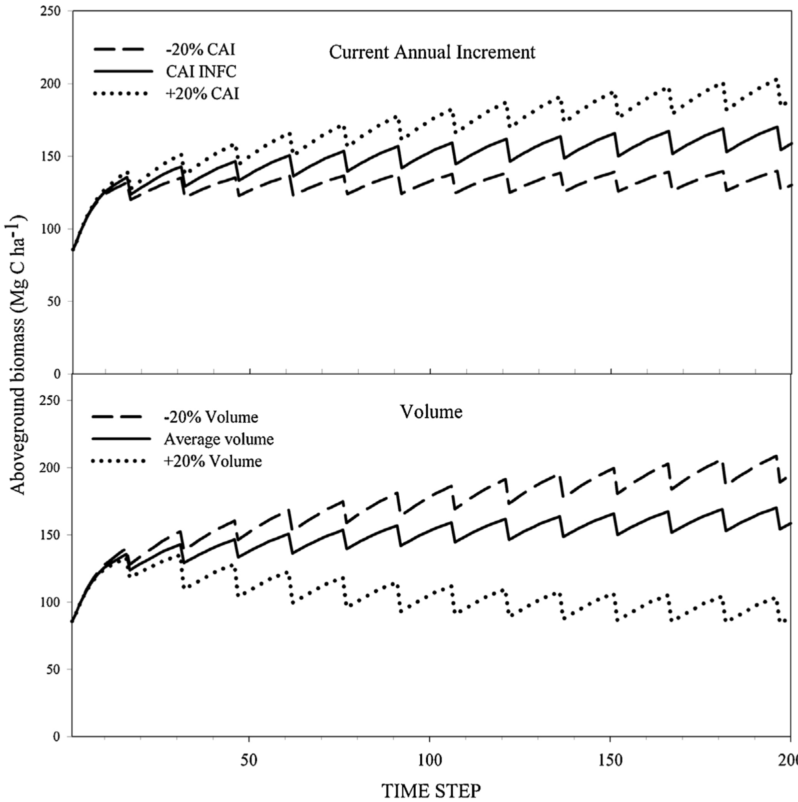

Fig. 7. Variation of the predicted total aboveground biomass due to a 20% reduction/increase of the average CAI and average volume reported by the INFC for spruce in the Trentino region.

causes a reduction of the percentage increment, which leads to a lower aboveground biomass. For both parameters, the effect was amplified by a longer simulation period. Thus, constant aboveground biomass could result for several stand characteristics: old forests with high volumes and slow growth, young forests with low volumes and fast growth, or forests with treatments maintaining a lower volume and fast growth. As highlighted by Fig. 8, the length of the cutting cycle also strongly affected the aboveground biomass stock.

The cutting cycle selected from the literature produced an equilibrium condition in the spruce and larch stocks. For fir and beech, the final C stock was not sustainable at the average cutting cycle (12 and 15 years, for fir and beech, respectively), which suggests that rotation lengths should be increased to maintain forest stocks. The

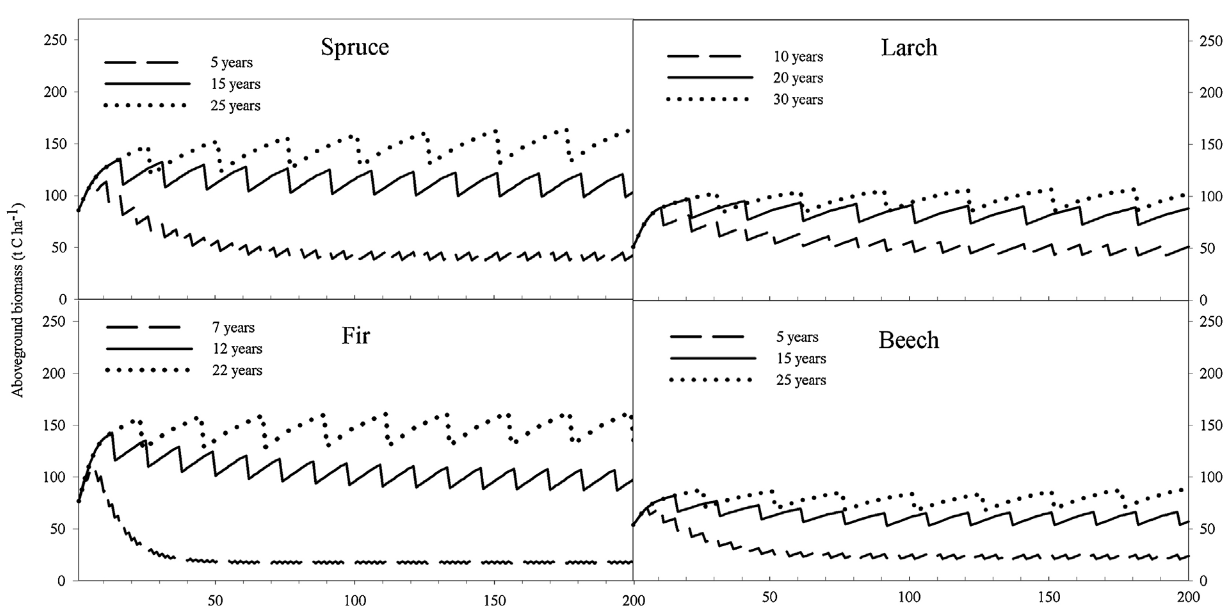

Fig. 8. Variation of the predicted total aboveground biomass for a spruce uneven-aged high forest in Trentino region, due to changes in rotation lengths for 20% commercial thinning events.

Table 3
C stock and stock change (average per unit of area and total for the country) for the period 1995–2009, including fire disturbance events.

	Pool	Dead organic matter		Soil	Living biomass		Total ecosystem
		Deadwood	Litter		Aboveground	Belowground	
C stock	Average ($Mg\,C\,ha^{-1}$)	9.1	8.0	56.0	53.1	12. 7	138.9
	Tot (Tg C)	68	60	417	395	94	1035
C stock change	Average ($Mg\,C\,ha^{-1}\,yr^{-1}$)	0.01	0.38	−0.47	0.56	0.12	0.68
	Total ($Gg\,C\,yr^{-1}$)	73	285	−349	4183	888	5080

other option of reducing the amount of biomass removed during the silvicultural treatment may be not economically viable. Shorter cutting cycles (between 10 and 5 years compared to the standard reported in the literature) always yielded a decreasing stock, while longer cutting cycles (+10 years compared to the standard reported in the literature) generally increased the aboveground biomass stock.

These simulations satisfied the general assumptions proposed for the uneven-aged stands, simulating (i) a faster (but decreasing) re-growth phase during the first period following the partial cut and (ii) a decreasing growth phase during the following years. In order to simulate the faster increment-phase following the disturbance event, the stand age was reset to 30 (i.e., the reference age class) immediately following a partial cut disturbance (Kull et al., 2011). The resulting pattern in Fig. 7 clearly corresponds to the growth model described by Hellrigl (1973) and by other studies (i.e., Tahvonen et al., 2010) for the uneven-aged high forest system. According to the general assumption proposed by Mayer (Bettinger et al., 2009), the current growth was periodically removed by routine partial cuts which maintained the initial volume.

Between 1995 and 2005, the uneven-aged forest area affected by harvest (i.e., some management practice) was equal to about 1000 kha over a total uneven-aged area equal to about 2157 kha (also including irregular and not classified forests). This figure is lower than the value reported by the INFC of about 1300 kha, including the forest area affected by single tree and group selection systems and other systems such as partial cuts on very small forest areas. This suggests that about half of the forest area in our study's "uneven-aged group" was not recently affected by harvest, which is different than previous theoretical assumptions (Schutz, 1997; O'Hara, 2001). Some of these forests could therefore evolve, in a long period of time, towards an even-aged forest structure, assuming that this last one is closer to the natural structure of many Italian forests (Cappelli, 1991; Del Favero, 2004). We did not consider this transition in age-class structure due to the short period of time covered by our analysis, but it could be simulated by applying the same approach that we proposed for the natural transition from coppices towards high forests (see Appendix A for further details).

3.3. Dynamics of C stock changes in the various pools

Throughout the simulations, Italy's forests were a net carbon sink. Table 3 reports the average C stock and C stock change estimated by the model for the period 1995–2009 based on the historical harvest rate, including the effect of fires. The total C density (stock per ha) averaged 139 $Mg\,C\,ha^{-1}$ resulting in a total C stock of 1035 Tg C, of which 47% is in the living biomass pools, 40% in the soil and the remaining 13% in the dead organic matter pool. The average soil C stock estimated in 2009, including the effects of fires, was 56 $Mg\,C\,ha^{-1}$, assumed to a depth of 1 m.

The total average annual C stock change was 0.68 $Mg\,C\,ha^{-1}\,yr^{-1}$ (82% provided by the aboveground biomass pools) and on the entire forest management area it was 5080 $Gg\,C\,yr^{-1}$.

During the same period, soil and dead organic matter represented, respectively, a small source and a small sink.

The C stock change between 1995 and 2009, based on the historical harvest rates (excluding fires), is reported in Fig. 9. The total C stock change (i.e., the sink in all C pools) increased from 4201 Gg $C\,yr^{-1}$ in 1995 to 6501 Gg $C\,yr^{-1}$ in 2009. The C stock change for biomass increased from 1995 to 2002, and then it slowly decreased in response to an increasing harvest rate, reaching 6845 Gg $C\,yr^{-1}$ in 2009. The C stock change for litter and dead wood (DOM) was negative in 1995 but decreased to −21 Gg $C\,yr^{-1}$ in 2009, showing a general positive correlation with variations in harvest rates. Soil C stocks decreased slightly (<1% yr^{-1}) throughout the simulation with a loss rate of −323 Gg $C\,yr^{-1}$ in 2009.

The total C stock change in 2020 ranged from 4707 Gg $C\,yr^{-1}$ to 6396 Gg $C\,yr^{-1}$, assuming an increased harvest rate (Increased-harvest-no-fire) and constant harvest rate (Constant-harvest-no-fire) after 2009. Based on the sensitivity analysis performed on the Increased-harvest-no fire scenario (also reported by Fig. 9), the total C stock change in 2020 ranged from 3740 Gg $C\,yr^{-1}$ to 5442 Gg $C\,yr^{-1}$ assuming a ±10% change in harvest rate.

As highlighted by Fig. 9 the biomass C stock change estimated by the model increased by about 30% from 1995 to 2009. Most of this increase occurred between 1998 and 2002, and was affected by a marked decrease in harvest rate (see Fig. 2).

Soil was a decreasing small source of C. This trend appeared to be not directly affected by harvest rates, but may reflect the transition from the disturbance regime assumed during initialization of the soil C pools and the actual disturbance regime that led to the initial age-class distribution at the start of the simulation. In contrast, the C stock change in litter and dead wood was primarily linked to changes in harvest rates because of the relatively high amount of residues transferred to the DOM pools during thinnings and in tree selection systems.

The addition of fire disturbances (Fig. 10), reduces the strength of the C sink in Italy's forests and introduces strong inter-annual variability in DOM and living biomass pools. These changes are directly related to the area annually burned. As expected, biomass and DOM pools showed an opposing pattern: fires kill trees and decrease the biomass C stock, but DOM C pools increase because the transfer of C to dead wood and litter adds more C than is lost from these pools during the fire. This effect was particularly significant in 2007, when the biomass C stock increase was reduced by about 3000 Gg $C\,yr^{-1}$ while DOM pools increased by about 2000 Gg $C\,yr^{-1}$ C. Between 1995 and 2009, fires decreased the total C sink by an average of 13% (ranging from 6% to 21% in specific years). The assumptions made to parameterize the fire disturbance matrix affect these results, and further information on the fuel consumption from DOM and living biomass pools during fires could improve our results.

Increased future harvest rates will also reduce the C sink strength in Italy's forest. The estimation of the forest C sink for the period 2010–2020 was based on two harvest rate scenarios: an increasing harvest (scenario Increased-harvest-no-fire: +36% compared to 2000–2009 average) or a constant harvest (scenario Constant-harvest-no-fire: average 2000–2009). The total C sink in 2020 was −17.3 Tg $CO_2\,yr^{-1}$ with the scenario Increased-harvest-no-fire and −23.5 Tg $CO_2\,yr^{-1}$ with the scenario Constant-harvest-no-fire. Comparing the various scenarios

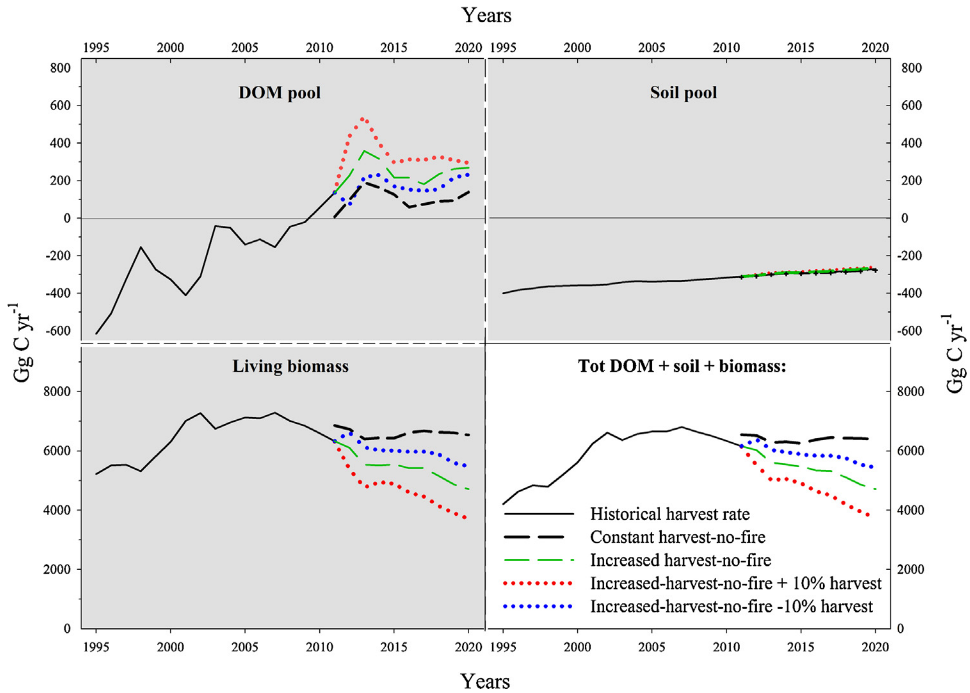

Fig. 9. C stock change (Gg C yr^{-1}) estimated for DOM (i.e., litter + dead wood), soil and living biomass pools, excluding fire disturbance events. The figure reports: (i) data based on the historic harvest rate (1995–2009); (ii) projections to 2020 based on the scenarios Increased-harvest-no-fire and Constant-harvest-no-fire; the sensitivity analysis on the scenario Increased-harvest-no-fire, assuming a ±10% variation of harvest demand, is also shown.

Table 4
Comparison between Scenario 1 (Increased-harvest-no-fire, including the sensitivity analysis) and Scenario 2a (Constant-harvest-no-fire). The last columns report (i) the percentage difference between the 2020 harvest demands applied to the scenario Constant-harvest-no-fire and to the other scenarios and (ii) the percentage difference on the 2020 total C sink.

Scenario	Net harvest demand in 2020 (million m^3)	2020 C sink (Gg CO_2 yr^{-1})	Comparison with constant-harvest-no-fire	
			Δ on harvest	Δ on C sink
Constant-harvest-no-fire	11.34	−23452	0%	0%
Increased-harvest-no-fire–sensitivity–10%	13.94	−19953	+23%	−15%
Increased-harvest-no-fire	15.49	−17259	+36%	−26%
Increased-harvest-no-fire–sensitivity +10%	17.04	−13714	+50%	−42%

and including the sensitivity analysis on the scenario Increased-harvest-no-fire (Table 4), the strong impact of harvest rates on the C sink emerges. Interestingly, this effect is not fully proportional, and depends on the initial harvest level: for each 1% increase in the 2020 final harvest demand, the C sink decreases by 0.75%, 0.86% and 1.05% assuming an initial 2020 harvest rate equal to 13.9 Mm^3, 15.5 Mm^3 and 17.04 Mm^3, respectively. Therefore, the effect of increasing harvest levels becomes stronger as the harvest removals approach the net annual increment. In contrast, with relatively lower harvest rates, the impact of other drivers (i.e., age structure, fires and natural mortality) becomes relatively more important.

The impact of management practices on the forest C sink dynamic has been investigated by several studies (Böttcher et al., 2008b). Böttcher et al. (2012) highlighted the role of the harvest rate as main driver for the future C sink in European forests by applying the EFISCEN and G4 M models to an increasing harvest demand scenario. The European Forest Sector Outlook Study II (UNECE/FAO, 2011b), which applied different policy scenarios to European countries, also highlighted the effects of the proportion of harvest derived from thinning, the rotation length and the amount of residue removals on the final C sink.

Compared to the total C sink estimated in the scenario Constant-harvest-no-fire, the inclusion of fire disturbances decreased the 2020 C sink by about 11% (scenario Constant-harvest-Min-fire) and 19% (scenario Constant-harvest-Avg-fire). Significant impacts of disturbance rates on the current and future C sink are also reported in other studies based on the CBM (Stinson et al., 2011; Metsaranta et al., 2010) or the EFISCEN (Schelhaas et al., 2002; Seidl et al., 2009) model.

The net balance of CO_2 emissions and removals (Gg CO_2 yr^{-1}) estimated by the model for each scenario is summarized in Fig. 11. According to IPCC reporting guidelines (IPCC, 2003), emissions are reported from the atmosphere's perspective such that negative values represent a terrestrial C sink (i.e., positive C stock change in the forest) and positive values a C source (i.e., negative C stock change in the forest). Total removals (including the effect of fires) varied between −14,542 Gg CO_2 yr^{-1} in 1995 and −21,381 Gg CO_2 yr^{-1} in 2009. The 2020 sink without fires ranged between −17,259 Gg CO_2 yr^{-1} and −23,452 Gg CO_2 yr^{-1} assuming, respectively, an increasing harvest rate and a constant harvest rate. The 2020 sink (excluding the impact of non-CO_2 emissions) ranged between −20,927 Gg CO_2 yr^{-1} and −19,091 Gg CO_2 yr^{-1} assuming a minimum and average fire level, respectively. The DOM pool increased in years with large fire events (i.e., in 1998 or 2007), and decreased in years with modest fires.

We estimated a slight reduction (about 0.21% per year) in the total C sink from −21,381 Gg CO_2 yr^{-1} in 2010 to −20,927 Gg CO_2 yr^{-1} in 2020 in the Constant-harvest-Min-fire scenario. The reduction in sink strength appears to be linked to an overall

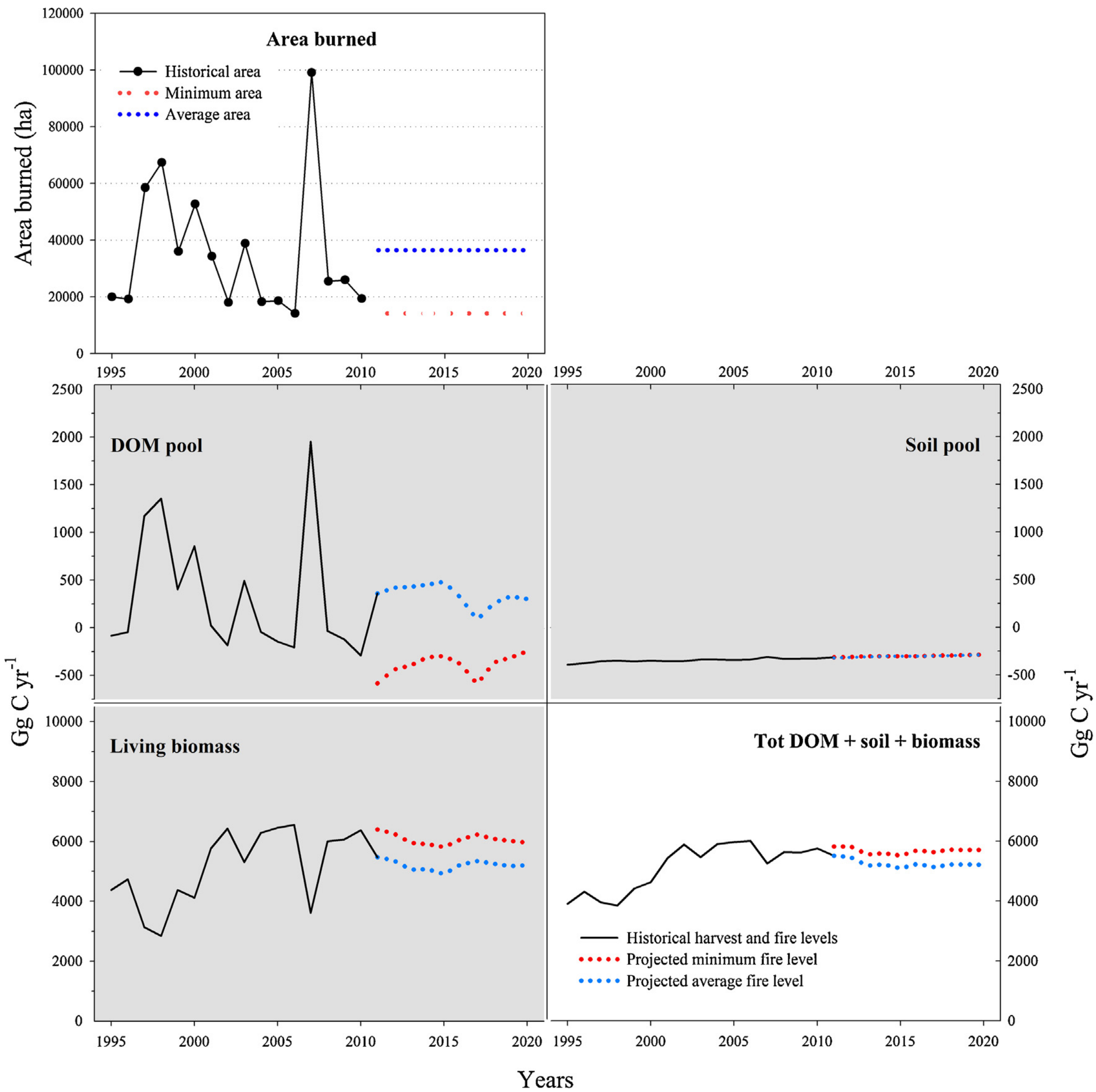

Fig. 10. C stock change (Gg C yr^{-1}) estimated by the model for DOM (i.e., litter + dead wood), soil, living biomass and total pools, considering historical and projected fire disturbance events. The figure reports: (i) until 2009, model results using the area historically disturbed by fires (combined with historical harvest rates); (ii) from 2010 onward, model results using the minimum (scenario Constant-harvest-Min-fire) and the average (scenario Constant-harvest-Avg-fire) level of historical burned area (combined with the scenario Constant-harvest-no-fire).
The top panel shows the area burned for the period 1995–2010, including the minimum (i.e., 14.1 kha yr^{-1}) and the average (36.4 kha yr^{-1}) areas used in scenarios Constant-harvest-Min-fire and Constant-harvest-Avg-fire.

decreasing CAI over time, as the average age of the forests in Italy increases. Indeed, the first Italian NFI (MAF-ISAFA, 1988 for the period 1983–1985) estimated an average aboveground biomass increment of 8.6 m^3 ha^{-1} yr^{-1} and 6.7 m^3 ha^{-1} yr^{-1} for even-aged and uneven-aged high forests, respectively (no data were provided by the first NFI on CAI for coppices). For 2005, INFC reports an average CAI about 30% lower than reported by the first Italian NFI (6.2 m^3 ha^{-1} yr^{-1} and 4.6 m^3 ha^{-1} yr^{-1} for even-aged and uneven-aged high forests, respectively). Even if the data provided by the two inventories are not fully comparable because of a different forest definitions, these differences suggest a decreasing trend of the CAI at least partially attributable to forest ageing, as observed also for other countries (Bellassen et al., 2011; Böttcher et al., 2012; Nabuurs et al., in press). The Italian situation, however, could be more complex because the tendency of an age-related decline of CAI in high forests could be partially compensated by the increase of CAI from the conversion of coppices to high forests (a process considered in our study and involving about 709,000 ha of coppices between 1995 and 2020).

3.4. Comparison with other studies

3.4.1. Soil

The calibration of DOM parameters was based on the comparison of the DOM C stock estimated by the model for the main DOM pools with data at the regional level, where comprehensive data were available on forest biomass and soil. In Fig. 12 the average soil C stock density (Mg C ha^{-1}) estimated by the CBM model is compared with the estimates reported in the literature at regional and national levels. CBM estimates of soil C are assumed to include the belowground slow pool and the belowground very fast pool to a depth of 1 meter.

For the Veneto and Trentino regions (high forests) our values fall into the confidence interval reported by other studies based on direct field measurements. Our estimates are also comparable with the data provided for the Piemonte and Tuscany regions and data estimated by the YASSO model at the national level (EFSOS2-EFISCEN results 2011–2012). In contrast, the overall C stock estimated by our model for Italy was (i) higher than the

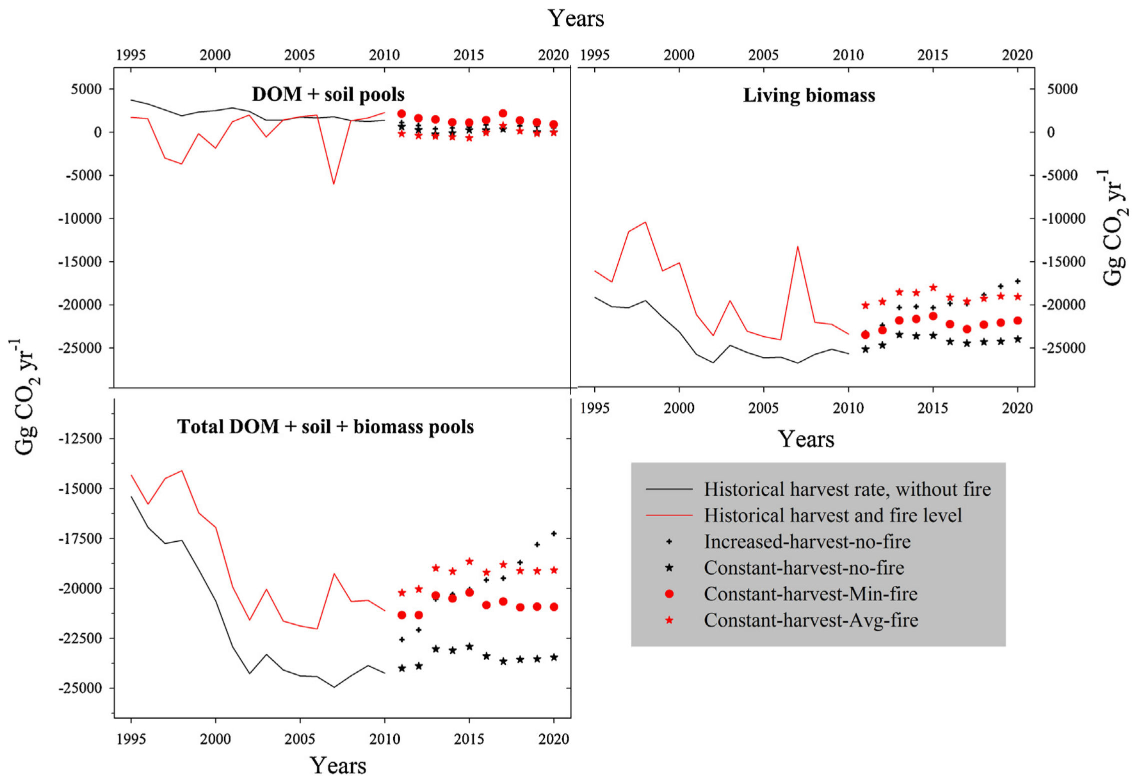

Fig. 11. Total C stock change (in Gg CO_2 yr^{-1}) estimated by model for DOM and soil pool, living biomass and the total pools, for each scenario from an atmospheric perspective where negative values represent a sink and positive a source (IPCC, 2006).

estimate reported by Liski et al. (2002), i.e., about 20 Mg C ha^{-1} (for 1990), (ii) about 38% lower than the C stock (76.1 Mg C ha^{-1}) recently reported by the INFC (Gasparini and Tabacchi, 2011). The value reported by Liski et al. (2002) was derived by a dynamic soil C model and, as highlighted by the same authors, it could underestimate the soil C stock because (i) it considered only the topmost soil layer (<20 cm depth), (ii) similar to the estimates provided by our model, it considered only the soil carbon originating from trees and (iii) it covered both the other wooded land and the forest land (i.e., for Italy a total area of 8550 kha).

The recently published value from the INFC was based on 1499 direct field measurements conducted during the period 2009–2010, for the mineral layers between 0 and 30 cm. The differences with our estimates can be related to (i) the total amount of forest area considered by the INFC (i.e., about 8759 kha, compared to 7450 kha included in our study) and (ii) the effect of the preliminary calibration of the soil and DOM parameters applied by our model, which was based on regional level data reported in the literature until mid-2011 (without INFC data). Overall, the values estimated by CBM fall within the high range of values reported by these regional studies (see Fig. 12). The wide range of existing estimates for soil C density in Italy's forests highlights the need for additional ground plot measurements.

Similarly to other soil models (UNECE–FAO, 2011a), the results provided by CBM are also influenced by the uncertainty in model initialization. For the initialization of DOM pools used by the CBM (Kurz et al., 2009), we assumed that the historic natural disturbance regime is a stand-replacing fire with a disturbance-return interval of 250 years. Other models such as the YASSO model applied by EFISCEN (Verkerk et al., 2011b), assume that the observed soil C pools are in a steady state where inputs equal losses through heterotrophic respiration. However, observed stocks may not be in equilibrium due to disturbances and very long turnover times of stable compounds (Wutzler and Reichstein, 2007). The initialization assumption of the CBM model reflects changes in disturbance regime at the start of the simulation relative to historical conditions, as explained above.

3.4.2. Litter and dead wood

The average value estimated by our model at the national level for litter, equal to 8.0 Mg C ha^{-1}, was higher than the value reported by Tonolli and Salvagni (2007) for the Trentino region, equal on average to 5.4 Mg C ha^{-1} with a coefficient of variation equal to 9%. The lower value reported by these authors could be related to some difference in the pool content. Indeed, Tonolli and Salvagni (2007) included in this pool leaves, small branches, cones, seeds and dead herbaceous vegetation. The CBM did not explicitly consider this last category but included in this pool 50% of the dead fine roots (<5 mm diameter).

The average value estimated by the model for deadwood, 9.1 Mg C ha^{-1} between 1995 and 2009, is equal to about 13% of the living biomass. This value includes four different components (Kurz et al., 2009; see Fig. 5): snag stems (1.8 Mg C ha^{-1}), snag branches (1.1 Mg C ha^{-1}), the medium pool (including the coarse woody debris on the ground) and the belowground fast pool (including dead coarse roots in the mineral soil, ≥5 mm). The total average volume estimated by INFC for the dead standing trees with a diameter ≥4.5 cm and for the downed deadwood with a diameter ≥9.5 cm is equal to 9.2 m^3 ha^{-1} (INFC, 2009). Assuming an average biomass and expansion factor of deadwood equal to 0.40 (Tabacchi et al., 2010), the dry biomass of this pool is equal to 3.68 Mg ha^{-1}, i.e., 1.84 Mg C ha^{-1}. This is almost the same value estimated by CBM for the snag stem pool

Our estimate can also be compared with the total average standing and downed deadwood reported for Italy by Verkerk et al. (2011a,b), based on the EFISCEN model combined with the soil model YASSO (EFSOS2-EFISCEN results 2011–2012). The value reported by these authors, equal, for 2010, to 3.9 Mg dry weight ha^{-1} (i.e., assuming a 0.5 content of C, 1.95 Mg C ha^{-1}) is about 45%

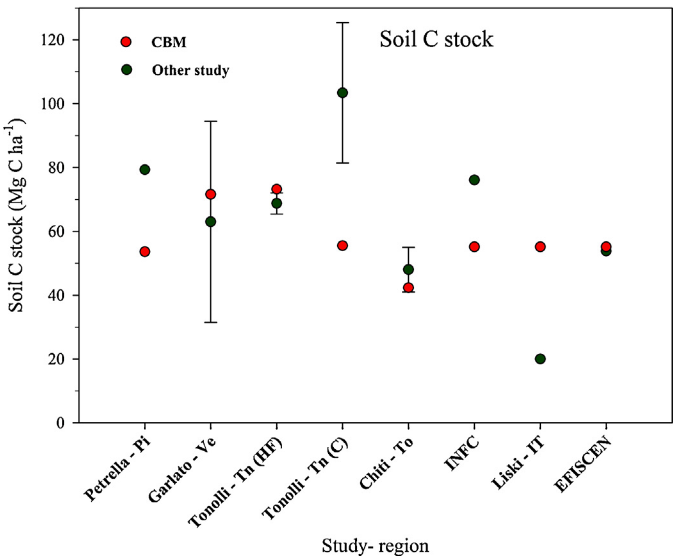

Fig. 12. comparison between the average soil C stock (in Mg C ha^{-1}) estimated by CBM model (assuming a depth of 1 m) and the following studies:

- Petrella and Piazzi (2005), Piemonte region (Pi): direct field measurements, depth 0–30 cm.
- Garlato et al. (2009), Veneto region (Ve): direct field measurements. depth 0–30 cm and coefficient of variation.
- Tonolli and Salvagni (2007), Trentino region (Tn): average high forests (HF) and coppices (C) C stock, direct field measurements, depth 0–30 cm (excluding the C content of the top most organic layer) and coefficient of variation. The figure reports the sum of the C stock of the three layers (i.e., the value is referred to a depth of 0–30 cm) and the total percentage coefficient of variation estimated by the values reported for the three layers, using the IPCC method for combining uncertainties (IPCC, 2003).
- Chiti et al. (2011), pure oak forest in Tuscany region (To): direct field measurements, depth 0–20 cm and standard error.
- INFC (Gasparini and Tabacchi, 2011), entire country: direct the field measurements, depth 0–30 cm (excluding the C content of the top most organic layer) and coefficient of variation (1.6%).
- Liski et al. (2002), entire country: dynamic soil C model.
- YASSO applied with the EFISCEN model (EFSOS2-EFISCEN results 2011–2012), depth of 0–20 cm.

lower than the total snags estimated by CBM model for the same year (i.e., 3.6 Mg C ha^{-1}). The EFISCEN model, however, did not consider the effect of fires that, according to our analysis, considerably increase the C stock of these pools. Other differences between the two models could be related to the stumps, to different mortality rates and to the amount of snags stems removed during thinning and clear cuts.

According to Hilger et al. (2012), the default snag fall rates applied by the CBM model and by our study, would be too low as compared with the snag fall rates derived from direct field measurements collected in Canada. This could cause an overestimation of the total dead wood C stock reported by our study.

3.4.3. *Biomass*

A detailed comparison between the results of our model and Tabacchi et al. (2010), who used data from INFC (i.e., based on direct field measurements), is possible for above ground biomass estimates in the year 2005 (i.e., the reference year for the INFC) and it is reported by Table 5.

Since Tabacchi et al. (2010) considered the total forest area (i.e., 8759 kha, including the forest expansion after 1990) and CBM used only forest existing in 1990 (i.e., 7450 kha), to allow for a direct comparison all values in Table 5 are expressed on an area basis. Furthermore, the original values from Tabacchi et al. (2010) were multiplied by 0.5 to convert dry matter to C.

The average aboveground C stock estimated in our study for the year 2005 was equal to 55.1 Mg C ha^{-1}. By excluding the leaves (i.e., 2.9 Mg C ha^{-1}), the resulting value (52.3 Mg C ha^{-1}) is slightly higher than the value estimated by the INFC (i.e., 50.5 ± 0.5 Mg C ha^{-1}) for the total Italian forest area.

Losses due to natural mortality in the CBM model were slightly lower (−16%) and the net growth was slightly higher (+1.4%) than the values reported by Tabacchi et al. (2010). However, the net growth reported by these authors already includes fires, while for CBM they were assumed as an additional disturbance event. Adding the losses due to fires to the value of net growth reported by Tabacchi et al. (2010) the resulting growth becomes 1.44 Mg C ha^{-1} yr^{-1}, i.e., slightly higher than the value estimated by our model (1.41 Mg C ha^{-1} yr^{-1}).

All CBM parameters fall into the 95% confidence interval of the parameters from the literature reported in Table 5, except for fire losses. Fire losses estimated by CBM (0.08 Mg C ha^{-1} yr^{-1}) were 60% higher than values reported by Tabacchi et al. (2010). This is due to the different assumptions made. First, our model used a larger burned area (18,159 ha) than the one used by Tabacchi et al. (2010), equal to 12,956 ha. In this latter study, it was assumed that 40% of the total 2005 burned area had been already affected by fires in previous years. Since this assumption is not explicitly supported by INFC data, we assumed that no forest affected by fire in each given year had been already burned in the previous years. Furthermore, Tabacchi et al. (2010) used the average volume per ha estimated by INFC for the high forests (i.e., 144 m^3 ha^{-1}) to estimate fire losses on the newly burned area. By contrast, the CBM model estimated fire losses based on the current stock of each forest type that was disturbed each year.

The total amount of harvest used by CBM model (13.0 Mm^3) is slightly lower than that used by Tabacchi et al. (2010) (13.8 Mm^3). However, when expressed on an area basis, our harvest (0.62 Mg C ha^{-1}) is 14% higher than the one used by Tabacchi et al. (2010), equal to 0.54 Mg C ha^{-1}. Further differences between the two studies can be related to the conversion from/to volume and biomass. Indeed, to compare the total amount of harvest provided by each model run and harvest scenario, we used basic wood density values for each species reported by the Italian NIR (Italy, 2011a). While these values may differ from those used by Tabacchi et al. (2010), directly derived from NFI field data, the impact of these differences on the final C sink is likely to be modest.

For the year 2005, Tabacchi et al. (2010) estimated a total C stock change for aboveground biomass of 14,700 Gg (with the 95% confidence interval ranging from 11,900 Gg to 17,500 Gg), equal to sink per ha of 0.83 ± 0.15 t C ha^{-1}. The corresponding sink estimated in our study, 0.72 Mg C ha^{-1} yr^{-1}, is about 13% lower than the value estimated by Tabacchi et al. (2010) but falls within its 95% confidence interval. This difference is explained by the higher losses due to fire disturbances (+0.03 Mg C ha^{-1} yr^{-1}) and to the higher amount of harvest per ha (+0.08 Mg C ha^{-1} yr^{-1}) estimated by the CBM model.

Overall, the differences between our study and Tabacchi et al. (2010) appear well explained by the different areas used. As compared to Tabacchi et al. (2010), we excluded both plantations and natural forest expansion after 1990 (in total, about 1300 kha for the year 2005). Since this young forest area likely has, per unit of area, a lower average biomass and a slightly higher sink than the forest area included in our study (see Italian NIR), it is logical that on an area basis the 2005 CBM estimate has higher biomass stock and a lower C sink than the estimates by Tabacchi et al. (2010).

A second study by Federici et al. (2008) allows for an additional comparison. Assuming an increasing forest area (9263 kha for 1990 and 11,144 kha for 2006) and excluding the effect of fire disturbance events, Federici et al. (2008) estimated the total biomass C sink for Italy was −21,597 Gg CO_2 yr^{-1} in 1990 (i.e., −2.33 Mg

Table 5
Comparison of the aboveground biomass pool estimates from CBM for 2005 (based on the scenario Constant-harvest-Min-fire) against the estimates directly detected by INFC or estimated by Tabacchi et al. (2010) on the basis of INFC measurements (all these values are reported with 95% confidence interval). Since the values reported in the literature referred to the entire Italian forest area (i.e., 8759 kha), results are reported as average C density (Mg C ha^{-1}).

Pool-Parameter	CBM	Literature	Source and comments on differences
Aboveground biomass stock	52.3 Mg C ha^{-1}	50.5 ± 0.5 Mg C ha^{-1}	[1] For CBM, values refer to biomass, excluding leaves.
Natural losses	0.10 Mg C ha^{-1} yr^{-1}	0.12 ± 0.04 Mg C ha^{-1} yr^{-1}	[2] The CBM estimate is the total stem and branch biomass transferred from living to DOM pools, excluding disturbance impacts.
Net growth	1.41 Mg C ha^{-1} yr^{-1}	1.39 ± 0.06 Mg C ha^{-1} yr^{-1}	[2] For [2] this value already included losses due to fires.
Fire losses to atmosphere	0.02 Mg C ha^{-1} yr^{-1}	0.05 ± 0.01 Mg C ha^{-1} yr^{-1}	[2] The total amount of burned area was equal to 12,956 ha for [2] and 18,159 ha for CBM. In [2] fires were assumed as natural disturbances.
Fire losses to DOM pools	0.06 Mg C ha^{-1} yr^{-1}		
Harvest to forest products	0.44 Mg C ha^{-1} yr^{-1}	0.54 ± 0.13 Mg C ha^{-1} yr^{-1}	[2], assuming a total demand equal to 13.8 Mm^3 (including logging residues). CBM assumed a total harvest demand equal to about 13.0 Mm^3 and an average amount of logging residues equal to 15% of the total demand.
Logging residues and transfer of C to DOM pools due to harvest disturbances	0.18 Mg C ha^{-1} yr^{-1}		
Aboveground biomass stock change	0.72 Mg C ha^{-1} yr^{-1}	0.83 ± 0.15 Mg C ha^{-1} yr^{-1}	[2] The differences in the final value are the result of net growth minus fire and harvest losses.
References	[1] → INFC (2009) [2] → Tabacchi et al. (2010): all values referred to aboveground biomass and reported as tons of dry matter. This was converted to C applying a 0.5 conversion factor to the reported values.		

CO_2 ha^{-1} yr^{-1}) and increased to −38,868 Gg CO_2 yr^{-1} in 2006 (i.e., −3.48 Mg CO_2 ha^{-1} yr^{-1}). The differences with our study, which estimated a less pronounced increase of the C sink for the same period, are related to (i) the forest area (assumed as a constant parameter in our study and as an increasing parameter by Federici et al. (2008); (ii) the amount of fellings, which increased in our study according to data provided by INFC; and (iii) more recent input data used by the CBM model. The study by Federici et al. (2008) estimated the C sink as a function of growing stock, applying a Richards function and comparing results with the CAIs provided by the first Italian NFI (MAF-ISAFA, 1988). As discussed above, these latter CAI values were generally higher than those reported in the more recent INFC, i.e., the average CAI reported by the INFC (and used in our study) was 4.1 m^3 ha^{-1} yr^{-1}, while the CAI used by Federici et al. (2008) was 6.3 m^3 ha^{-1} yr^{-1}.

A third comparison may be done using the values reported by Italy in its submission on forest management reference levels (Italy, 2011b see Fig. 13). Since the soil was not considered in the submission, this pool is also excluded from the CBM results reported in this figure. With the exception of one year (2007, characterized by large fires), the C sink estimated by our model is on average about 30% lower than the C sink reported in Italy's reference level submission. Furthermore, our results show a lower inter-annual variability. Most of this difference may be explained by the fact that Italy based its estimate on the 1985 NFI, whose increment is about 30% higher (at least for the high forests) than the increment applied in our study (see above). The lower variability is mainly due to the handling of fires. In the CBM model, a fire kills biomass and transfers C to the dead wood C pool (e.g. see the year 2007 in Fig. 12) where it will slowly be released through decay. By contrast, Italy estimated the dead wood pool through a linear regression with the aboveground biomass (Italy, 2011a), so that a reduction in biomass C pool due to a fire causes a corresponding reduction in the dead wood pool which represents an immediate release to the atmosphere. This assumption creates larger inter-annual variability in Italy's reference level submission, in comparison to our results.

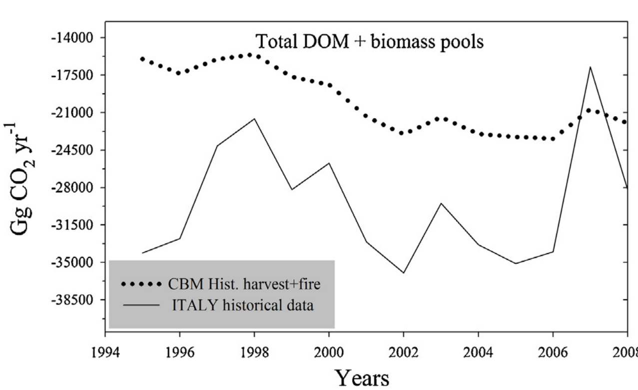

Fig. 13. Comparison of total litter, dead wood and living biomass C stock change (in Gg CO_2 yr^{-1}) estimated by the model based on the historical harvest and fire levels and the values reported by Italy in the last submission of information on forest management reference levels (Italy, 2011b). The soil pool is not included.

A fourth comparison is with results from the EFISCEN model. According to the reference scenario provided in the EFSOS II study (UNECE/FAO, 2011b), based on the current silvicultural practices and a total amount of fellings equal to 10.3 Mm^3, EFISCEN predicted a biomass C stock change for Italy equal to 0.50 Mg C ha^{-1} yr^{-1} in 2010. This value is about 42% lower than the value estimated by CBM model for the same year (i.e., 0.71 Mg C ha^{-1} yr^{-1}). The differences are probably related to uneven-aged forests and to different assumptions about management practices. Indeed, the current version of the EFISCEN model is particularly suitable for even-aged forests, while it is recognized that results are less reliable for the uneven-aged forests which represent about 30% of the FM area in our study, as well as for forests treated with shelterwood system (Verkerk et al., 2011a; UNECE/FAO, 2011b) which is commonly applied to beech forests in Italy that represent 12% of the area.

A final comparison may be done for net primary production (NPP, i.e., the sum of all biomass C production during a year, Kull et al., 2006). The average NPP estimated by our model between 1995 and 2009 (458 g C m^2 yr^{-1}, scenario Constant-harvest-Avg-fire) is slightly lower than the 510 g C m^2 yr^{-1} reported for Italy by Tŭpek et al. (2010) based on the EFISCEN model. This study, however, did not account for fires and other natural disturbance events, and was based on data provided by the first Italian

NFI (1985). In contrast, our value is higher than the NPP estimated by other models reported by the same authors, such as BIOME-BGC (401 $g\,C\,m^2\,yr^{-1}$), JULES (437 $g\,C\,m^2\,yr^{-1}$) or ORCHIDEE (333 $g\,C\,m^2\,yr^{-1}$), which are mainly based on the modeling of biochemical processes and typically do not include forest management activities.

4. Conclusions

The objective of our study was to use CBM and the latest NFI data to estimate the forest C dynamics in a country (Italy) exemplifying most of the complex and varying silvicultural systems applied in Europe. To this aim, after having addressed a number of methodological challenges, we performed an extensive evaluation of the model's results and then projected the forest C dynamics to 2020 under different harvest and fire scenarios.

The first challenge we faced was on the correct use of NFI parameters, namely the CAI, which represents the gross volume yield of each stand, and the standing volume, which reflects the net standing volume, including the impacts of past silvicultural activities such as thinning. This is a relevant issue for the application of CBM and potentially of other yield-data driven models to European countries. To address this issue, we used two yield curve libraries: a historic YT library based on NFI volume data to obtain the standing volume at the start of the simulation (1995), and a current YT library based on the CAI for the model runs to 2020.

A second challenge was estimating the forest C dynamics for a period antecedent to the reference NFI year (in our case, 2005). We reconstructed the 1995 age class structure for Italy's forests which allowed the validation of model results, through comparisons with historical data from other sources, mainly derived from the last NFI. Such comparisons showed that our estimates are largely consistent with other studies and, where differences emerged, these were explained by different assumptions and input data.

A third challenge was related to the fact that yield data-driven models, like CBM, cannot be directly applied to uneven-aged forests, where no yield tables are available. To overcome this limitation, we developed a novel approach based on volume and increment data provided by NFI for the uneven aged forests, and we adapted the default model design to the tree selection system. Since uneven-aged forests cover about 30% of the forest area in Europe, addressing this issue is relevant for the potential future application of the CBM to other countries.

In conclusion, our study demonstrated that the CBM can be successfully applied to simulate the recent and projected future forest C dynamics of European forests characterized by complex silvicultural systems. This study provides the foundation for the application of the CBM to other European countries. Further studies will test the use of this model to simulate forest land-use changes, additional disturbance types and different climatic conditions.

Disclaimer

The views expressed are purely those of the authors and may not in any circumstances be regarded as stating an official position of the European Commission or Natural Resources Canada.

Acknowledgments

We thank especially Prof. B. Hellrigl of the University of Padova for his help in defining the silvicultural strategy applied at national level; Giovanni Tabacchi and Patrizia Gasparini of the National Council for Agricultural Research for helpful comments and suggestions for improving this paper. We thank Stephen Kull, Scott Morken, Greg Rampley and the Carbon Accounting Team for their indispensable technical support during the study. We thank three anonymous reviewers who provided useful suggestions to improve the manuscript.

Appendix A. Calibration of the harvest demand and silvicultural systems

In the CBM model, harvest activity can be defined by the area, the proportion of eligible area to disturb, or by the amount of merchantable C to be harvested, i.e., transferred out of the forest to the forest product sector (Kull et al., 2006). In this study, harvest was applied as a clear cut area target (with corrections based on harvested volumes) and a thinning merchantable C target, as explained below. To separate the harvest by forest types (assumed as representative of the main species), we first subtracted the amount of merchantable C provided by clear cut in even-aged high forests and coppices from the total harvest demand. We assumed that the area affected by clear cuts between 1995 and 2004 was equal to the total forest area reported by INFC for the youngest age class (i.e., <10 years old). This value (13.38 kha) was equally distributed between 1995 and 2004 (i.e., 1380 $ha\,yr^{-1}$) and assumed constant until 2010. The annual harvest (equal to 0.08% of the even-aged high forest area) mainly included oaks, other broadleaved species and pines.

A more complex treatment, the shelterwood system, is suggested for beech even-aged high forests. In this case, a new age class develops beneath the moderated microenvironment provided by the residual trees. The sequence of treatments generally includes three distinct types of cutting: (i) an optional preparatory cut to enhance conditions for seed production; (ii) an establishment cut to prepare the seed bed and create a new age class; (iii) a removal-cut to release the established regeneration from competition of the residual overstory trees. This system, generally applied to beech high forests (Nocentini, 2009), was simulated as a 15% reduction in biomass applied every 15 years between 20–35 and 80–105 years (depending on the region), followed by a 30% reduction in biomass at 100–120 years (i.e., the preparatory cut) and by a clear cut at 120–140 years (i.e., the removal cut). Because the new age class should already be established before the removal cut, the age of the new forest, after the disturbance event was set to 10 years (Kull et al., 2006).

The simple coppice system and the coppice with standards system, where a certain number of standards (between 60 and 100 trees per ha) are retained (Coppini and Hermanin, 2007; Nocentini, 2009), were simulated as a clear cut harvest. Based on the age structure reported by INFC, 11,234 ha per year (equal to 0.3% of the total coppice area) were affected by clear cut between 1995 and 2005. Assuming also that other silvicultural treatments on very small areas (i.e., not detected by INFC) could affect these forests, we also applied a 15% removal of the merchantable biomass on coppices older than 25 years.

The selective coppice system (i.e., uneven-aged coppices) applied to beech and chestnut, where shoots of different ages (usually three age classes) grow on each stool and the biggest shoots are cut every 8–12 years (Del Favero and Lasen, 1993) was simulated by a 30% reduction in biomass applied every 10–12 years. The area affected by the selective coppice system was estimated by a calibration process together with the uneven-aged forests.

Coppices older than 50–60 years (depending on local conditions and FTs), generally undergo a process of conversion to high forests (Nocentini, 2009). The conversion can be actively managed by specific silvicultural treatments or can be left to a natural self-thinning in abandoned stands. In the first case, the stand density is progressively reduced by repeated thinning of the shoots, to enhance the growth of the best shoots and to reduce re-sprouting. In the second and most common case, the ageing process of the coppice results in a phase of high natural mortality, followed by a re-growth phase.

Because no information about the extent of transition from coppice to high forest was reported in the literature, all coppice stands older than 60 years at the start of the simulation were assigned to the high forest MT. During model simulations all coppice stands that reached 60 years transitioned to the high forest MT. This transition was associated with a general '5% disturbance' event (i.e., a disturbance that caused mortality of 5% of the aboveground biomass) to simulate the higher natural mortality observed in stands undergoing natural conversion.

The capability of the model to simulate the specific silvicultural systems applied to the even-aged forests was tested through detailed preliminary analysis (not reported in this manuscript).

Based on current silvicultural practices, we defined a further proportion of harvest provided by thinnings, simulated by an increasing percentage removal of the merchantable biomass applied every 10 years to all even-aged forests. Removals were 10% in stands between 10 and 30 years, 15% between 35 and 95 years and 20% over 100 years. This amount was distributed between FTs and regions according to the total proportion of aboveground C stock and defined as amount of merchantable C requested from the model.

Following the previous assumptions, the total annual amount of harvest defined at the national level was split between different regions and FTs based on (i) the age structure reported by INFC (used to estimate the clear cut amount), (ii) additional assumptions on the amount of thinnings (for even-aged forests) and (iii) the total C stock available for each stand, according to the output provided by the model (used to split the remaining harvest demand between different uneven-aged forest types).

The harvest demand not provided by even-aged forests was allocated to uneven-aged forests. It was distributed according to the proportion of aboveground C stock available at each step of the simulation, for each FT and region, and it was defined through the amount of area to be disturbed per each year. This allowed us to (i) easily apply the assumptions used to simulate the uneven-aged silvicultural system and to (ii) calibrate the harvest area as a function of the wood supply area provided by the INFC.

The harvest volume provided by uneven-aged forests was estimated from the amount of C reported by a preliminary model run as "merchantable wood products" (distinguished between hardwood and softwood, as reported by Table D.1) by:

$$V_n^i = Mwp_n^i \times \frac{1}{0.5} \times \frac{1}{DB_n} \tag{A.1}$$

where V_n^i was the merchantable volume in m^3, for each FT (subscript n) and region (superscript i) Mwp_n^i was the merchantable wood products in t of C, 0.5 was the carbon content and DB_n was the basic wood density (Italy, 2011a).

The volume estimated by Eq. (A.1) was compared with the data estimated for 1995, i.e., the starting point of our simulations. The area available for wood supply was then re-calibrated using the proportion of removals reported by the statistics for the same year, according to:

$$AA_n^i = \frac{V\_stat_n^i}{V\_CBM_n^i} \times AA\_\inf c_n^i \tag{A.2}$$

where AA_n^i was the re-calibrated area available for wood supply (expressed as percentage of the total area), $V\_stat_n^i$ and $V\_CBM_n^i$ were the merchantable volume based on the data provided by corrected national statistics and on the CBM simulation, respectively, for the same year, and $AA\_\inf c_n^i$ was the area available for wood supply based on the data reported by the INFC.

This approach was repeated, each time re-calibrating the annual area available for wood supply, until the difference between the volume estimated by the simulation and the value assumed by the national statistics was less than 2.5%.

Appendix B. Even-aged theoretic evolution of the CAI derived by the model and provided by original INFC data

Fig. B.1 and Fig. B.2 report the theoretic evolution of the CAI for even-aged high forests and coppices, respectively, based on the YTs derived by Eq. (3). In the same figures, this parameter is also compared with values reported by the INFC, highlighting that for all FTs the values provided by the current yield tables adequately represent the figures reported by the INFC.

Appendix C. Sensitivity analysis on parameters applied to uneven-aged growth curves

To estimate parameter b in the exponential function in Eq. (6), we performed a sensitivity analysis by varying b between 0.91 and 0.99 in increments of 0.01. Comparing the average increment reported by INFC for the uneven-aged FTs with the values reported by the previous Italian NFI (MAF-ISAFA, 1988), we found that the increment decreased, on average, 15–30% over a period of 20 years, or about 0.75–1.5% per year, depending on the FT. Due to the methodological differences between the two inventories, we cannot quantify this amount more precisely. The reduction of the increment can also be related to the effect of harvest and fires and to afforestation because new forests were mainly included in this group by INFC. Thus the reduction of 0.75–1.5% per year represents the maximum reduction that can be expected for the uneven-aged forests. For this sensitivity analysis, we estimated the average increment for each region over time using 10-year age classes for 6 FTs and a range of b values between 0.91 and 0.99 (see Fig. C.1). A value of b equal to 0.98 provided the best agreement between the two inventories and showed a gradual decrease of Ip_t, over about 60 years. Each line reported in Fig. C.1 represents the theoretical evolution of the CAI, excluding thinning or other disturbance events. We assume that when a selective cut occurs, the area affected by thinning is always transferred back into the first age class which has the same increment reported by INFC.

Our model assumptions on the uneven-aged high forests, were further validated by comparing, for each FT, the average merchantable volume estimated by the model for 2005 with the volume reported by the INFC for the uneven-aged and irregular management types, for stem and main branches (Fig. C.2). The average volume estimated by our model ($216\,m^3\,ha^{-1}$) was 8% higher than the average volume reported by the INFC ($200\,m^3\,ha^{-1}$). The volumes estimated at FT level, were generally higher than the figures reported by the inventory. These differences could be related both to the yield tables applied to these forests and to different assumptions on the merchantable volume compartment.

Appendix D. Calibration of stand-level equations

Stem wood biomass of the merchantable trees (i.e., the trees considered in the yield tables) was estimated as:

$$b\_m = aV^b \tag{D1}$$

where V = the gross merchantable volume of all live trees (excluding stumps, tops, or trees with a dbh< merchantable dbh), in $m^3\,ha^{-1}$, b_m = total stem wood biomass of the merchantable live trees (including stumps and tops), in $m^3\,ha^{-1}$, and a, b = non-linear model parameters fitted separately by province, ecozone, and main tree species.

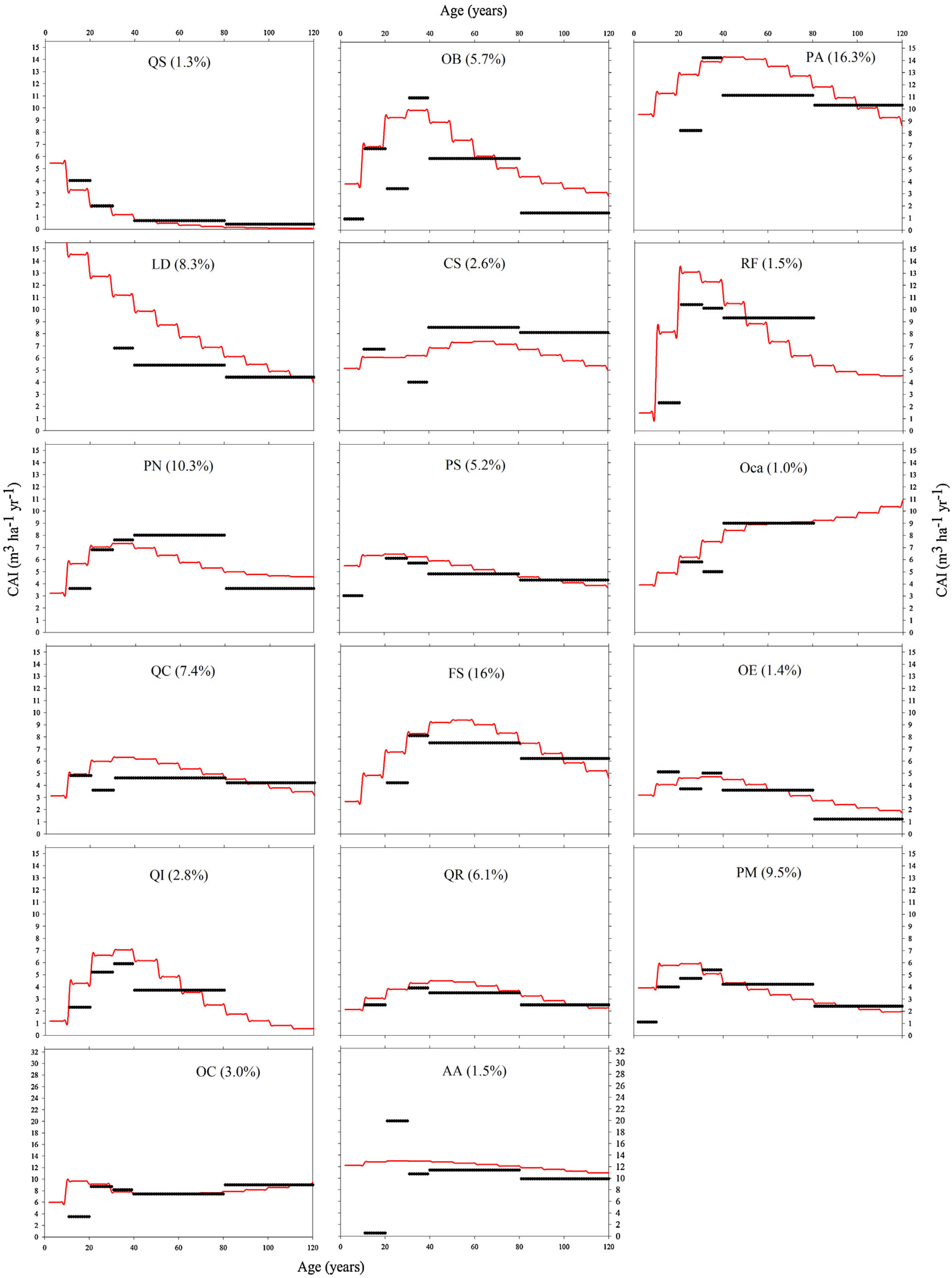

Fig. B.1. Average CAI (m^3 ha^{-1} yr^{-1}) estimated for each FT (red lines) based on the yield tables derived by Eq. (3). The figure reports the theoretic evolution of this parameter on undisturbed even-aged high forests and the average CAI inferred by INFC for each FT (black lines) according to the following age-class distribution: 0–10 years, 11–20 years, 21–30 years, 31–40 years, 41–80 years and 80–120 years. For some FTs and age classes, where no forests were detected by INFC, no data were provided by the inventory. FTs are reported according to acronyms listed in Table 1 with the share of area covered by each FT (expressed as percentage on the total even-aged high forest area reported by INFC). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

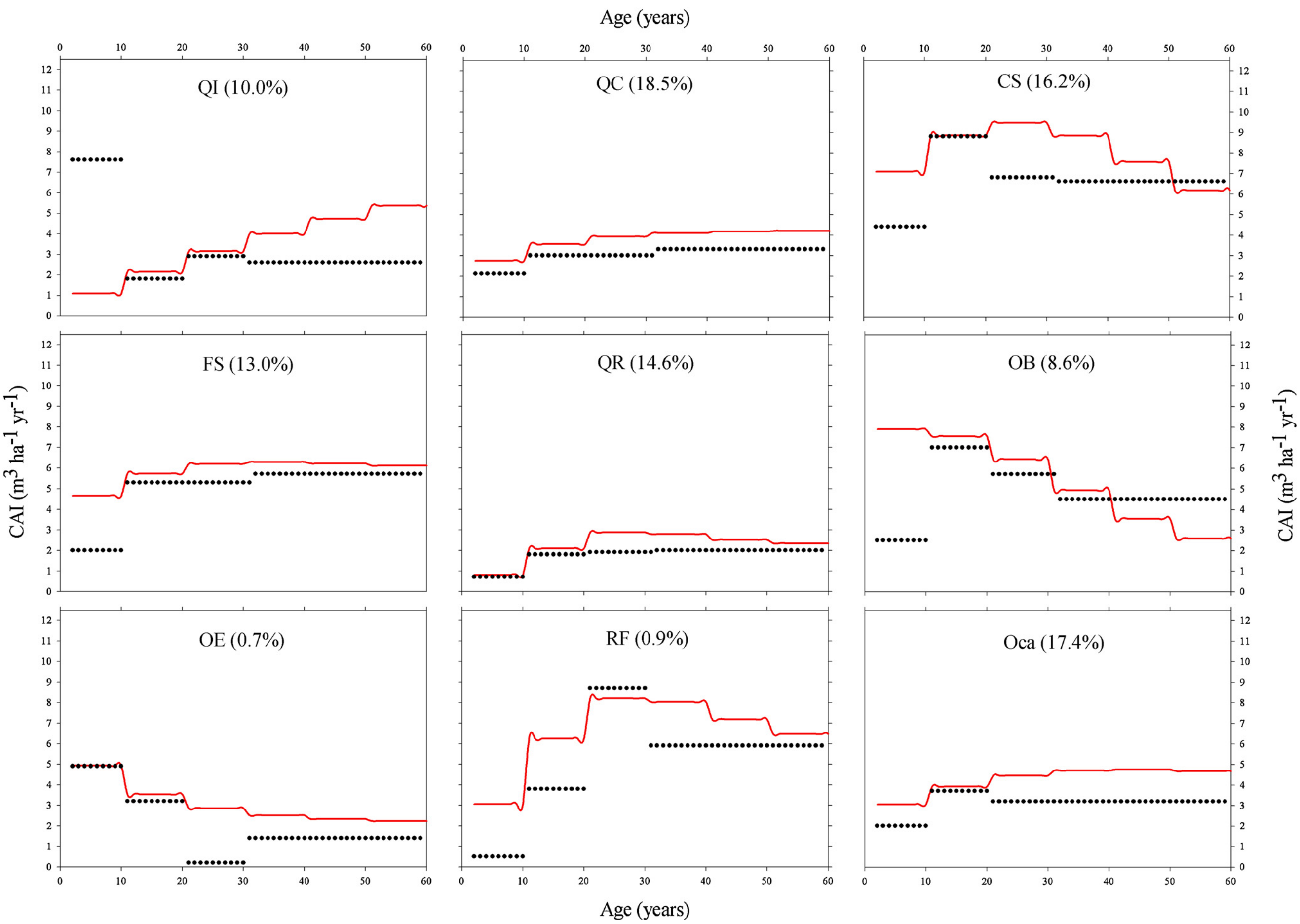

Fig. B.2. Average CAI ($m^3\ ha^{-1}\ yr^{-1}$) estimated for each FT (red lines) based on the yield tables derived by Eq. (3). The figure reports the theoretic evolution of this parameter on undisturbed coppices forests and the average CAI inferred by INFC for each FT (black lines) according to the following age-class distribution: 0–10 years, 11–20 years, 21–30 years, 31–40 years and 41–60 years (above this age coppices were converted to high forests). FTs are reported according to acronyms listed in Table 1 with the share of area covered by each FT (expressed as percentage on the total area, considering only coppices younger than 60 years according to the 1995 age class distribution). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

To predict the proportion of the total tree biomass in stem wood, bark, branches, leaves and stumps, the specific multinomial models used by Boudewyn et al. (2007) were applied. The resulting equations were used to estimate these compartments, e.g., the proportion of total tree biomass in stem wood, bark, branches, and stumps for each species. The volume of non-merchantable trees was estimated through additional expansion factors that quantify the amount of stem wood biomass contained in non-merchantable trees. Because the equations developed by Boudewyn et al. (2007) were based on data collected in Canada, the application of the default parameters provided by the CBM could overestimate or underestimate the wood biomass of other countries.

To test the fitness of these equations and to choose species-specific parameters suitable for Italy, the default database provided for the province of Quebec (where about 190 stand-level equations were available) was applied using the following steps:

- Applying as independent variable in each stand-level equation provided by Boudewyn et al. (2007) the average volume (V_{INFC}) reported by the INFC for each Italian administrative region and forest type (INFC, 2009), the following parameters were estimated:
  a. *AB_b*: total aboveground biomass, directly comparable with the mean total aboveground biomass reported by the INFC for trees with a Dbh > 4.5 cm;
  b. *SBr_b*: biomass of stems + branches (excluding tops, stumps and leaves), assumed as comparable to the biomass of stem and main branches (with diameter > 5 cm) reported by the INFC.
  c. *Stu_b*: biomass of stumps (estimated as a proportion of the merchantable stem biomass), directly comparable with the biomass of stumps reported by the INFC.
- The volume estimated for the smallest and non-merchantable trees could not be compared with specific data provided by the literature for Italy.
- For each forest type, administrative region and biomass component (i.e., *AB_b*, *SBr_b* and *Stu_b*) the sum of squares were calculated of the differences (*ss*) between the values predicted by the stand-level equations and observed by the INFC.
- For each forest type and component, the mean of the sum of squares was estimated, i.e., ss_{AB}, ss_{SBr} and ss_{Stu} for total aboveground biomass, biomass of stem + branches and biomass of stumps, respectively (see Tab D1).
- For each forest type and equation we estimated the average sum of squares ($\overline{ss}$) applying to each biomass component a weighting factor (w) between 1 and 0.02 and the following equations:

$$\overline{ss} = \frac{ss_{AB} \times w_{AB} + ss_{SBr} \times w_{SBr} + ss_{Stu} \times w_{Stu}}{w_{AB} + w_{SBr} + w_{Stu}} \tag{D2}$$

where

w_{AB} = weighting factor attributed to total aboveground biomass, equal to 1;

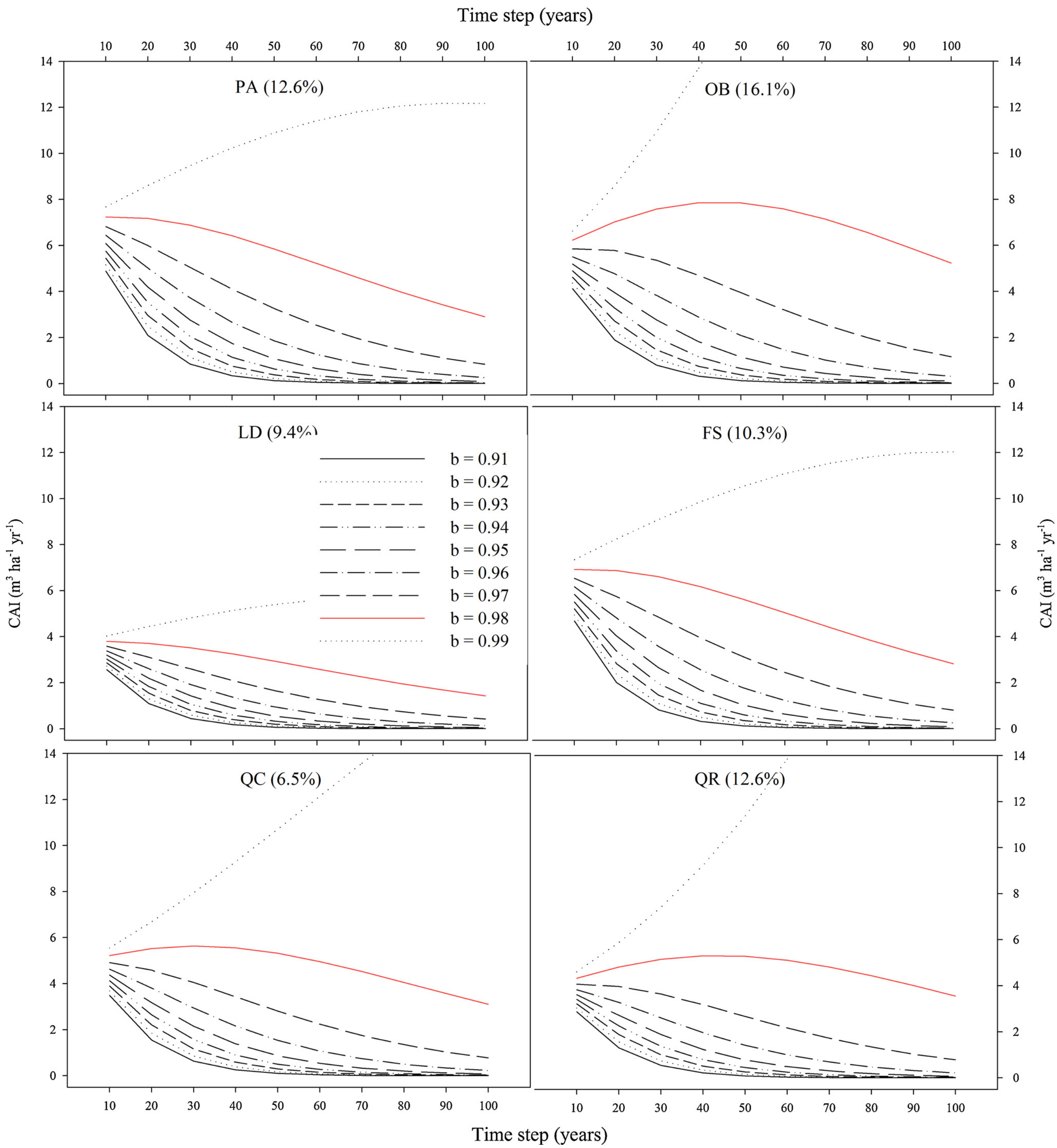

Fig. C.1. The theoretical evolution of the CAI ($m^3\ ha^{-1}\ yr^{-1}$) against annual time step for the 6 main uneven-aged FTs, based on different values of b (from 0.91 to 0.99). The CAI of each FT is the average of the increments estimated for each region. A value of b equal to 0.98 was used in this study (red line). The 6 FTs are listed by acronym (see Table 1) and their percentage of the total area of the uneven-aged is indicated in parenthesis. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

w_{SBr} = weighting factor attributed to biomass of stem + branches, estimated, for each forest type, as the average proportion of this component according to the values reported by the INFC;

w_{Stu} = weighting factor attributed to biomass of stumps, estimated, for each forest type, as the average proportion of this component according to the values reported by the INFC.

- For each forest type the equation that minimizes $\overline{ss}$ was selected. Using this equation the mean percentage difference was also calculated between the values of total aboveground biomass estimated and reported by the INFC for each forest type.

For each forest type and component, the average of the sum of squared error was estimated, i.e., ss_{AB}, ss_{SBr} and ss_{Stu} for the total aboveground biomass, biomass of stem plus branches and biomass of stumps, respectively (Table D.1).

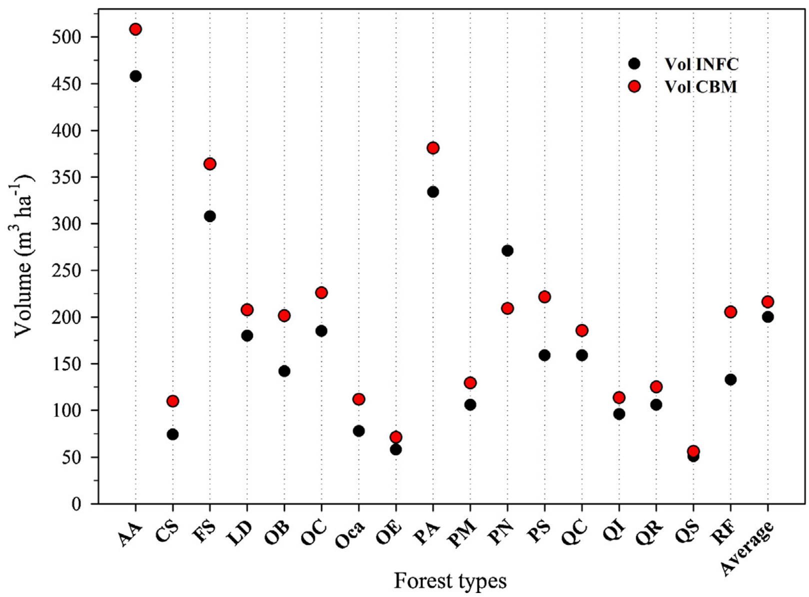

Fig. C.2. Comparison between the merchantable volume estimated by CBM for the uneven-aged FTs (referred to 2005) and the volume reported by the INFC for the uneven-aged and irregular FTs.

The percentage differences between the total aboveground biomass estimated through the selected equations and the biomass reported by INFC are reported in Fig. D.1.

Each Italian forest type was therefore linked to a stand-level volume to biomass equation originally developed for a default species in Quebec. A customized version of the *Archive Index Db* used to store default data for CBM projects was developed. This database contained the administrative and climatic parameters defined for Italy, the parameters of the equations selected to convert stand-level volume to biomass, and many other parameters used in model runs.

For nine forest types out of 19 (i.e., broadleaves and conifer plantations, other conifers and other evergreens, spruce, Mediterranean pines, Black and Scots pine and riparian forests) the equation for Red spruce (*Picea rubens*) produced the minimum error and its volume to biomass coefficients were selected. For the Holm oak category, the selected species, Butternut (*Juglans cinerea*), was considered a rare species whose minimum volume (65 m^3 ha^{-1}) was too large to adequately represent this forest type, and so the selection was replaced with White elm (*Ulmus laevis*) which had the second lowest $\overline{ss}$ value. The same species was also selected for the Cork oak category, while each of the remaining forest categories was associated with different species. The mean percentage difference between the total aboveground biomass estimated using the selected stand-level equations and the biomass reported by the INFC was 3.8% (Fig. D.1) and ranged between +12% for Mediterranean pines and −3% for the other evergreen forest type. This indicates that, overall, the use of stand-level equations originally developed for Quebec for the conversion of stand-level volume to component biomass estimation is adequate to represent the data provided by the INFC for each FT and region in Italy, although further reductions in uncertainty may be possible.

Table D.1
Sum of squares and weighting factors for total aboveground biomass (ss_{AB} and w_{AB}), stem + branches (ss_{SBr} and w_{SBr}) and stumps (ss_{Stu} and w_{Stu}); minimum average-weighted sum of squares ($\overline{ss}$) estimated by the Canadian volume to biomass equations and name of the selected species. The last column distinguishes the forest type of the selected species as either Hardwood (HW) or Softwood (SW).

Forest categories	Number of obs.	Sum of squares			Weighting factors			Average	Selected species (Quebec)	Type
		ss_{AB}	ss_{SBr}	ss_{Stu}	w_{AB}	w_{SBr}	w_{Stu}	$\overline{ss}$		
AA	14	2399.4	1290.0	6.2	1	0.77	0.02	1896.3	White spruce	SW
CS	21	619.7	615.7	2.4	1	0.76	0.02	609.7	Balsam poplar	HW
FS	20	3163.6	2983.1	5.3	1	0.78	0.02	3043.1	Gray birch	HW
LD	8	403.0	34.0	3.9	1	0.83	0.02	233.4	Eastern white-cedar	SW
OB	21	618.3	303.3	0.9	1	0.73	0.02	479.0	Eastern white pine	SW
OC	17	1054.3	1889.9	15.3	1	0.74	0.02	1394.8	Red pine	SW
OE	9	47.9	27.9	0.0	1	0.75	0.02	38.8	Red pine	SW
Oca	19	290.5	138.3	2.3	1	0.70	0.03	224.6	Black spruce	SW
PA	13	1147.6	2564.1	16.9	1	0.76	0.02	1742.7	Red pine	SW
PM	17	1088.8	2235.8	10.4	1	0.74	0.02	1561.9	Red pine	SW
PN	21	2681.1	1529.5	12.3	1	0.71	0.02	2179.4	Red pine	SW
PS	12	637.8	1150.1	8.5	1	0.75	0.02	849.4	Red pine	SW
QC	16	723.0	472.6	1.5	1	0.78	0.02	605.4	Largetooth aspen	HW
QI*	16	686.4	79.1	1.5	1	0.72	0.01	428.4	Butternut*	HW
QR	21	309.3	415.7	0.9	1	0.73	0.02	350.1	Basswood	HW
QS	6	89.6	218.8	0.8	1	0.76	0.02	144.0	White elm	HW
RF	21	509.1	1234.1	1.0	1	0.73	0.02	803.2	Red pine	SW
* for this forest type, the selected species, Butternut, considered as a rare species whose minimum volume provided by the stand-level equation model (65 m^3 ha^{-1}) cannot adequately represent the *Quercus ilex* category, the selection was replaced with the second species, having the minimum ss value:										
QI	16	272.7	1255.6	7.9	1	0.72	0.01	678.9	White elm	SW

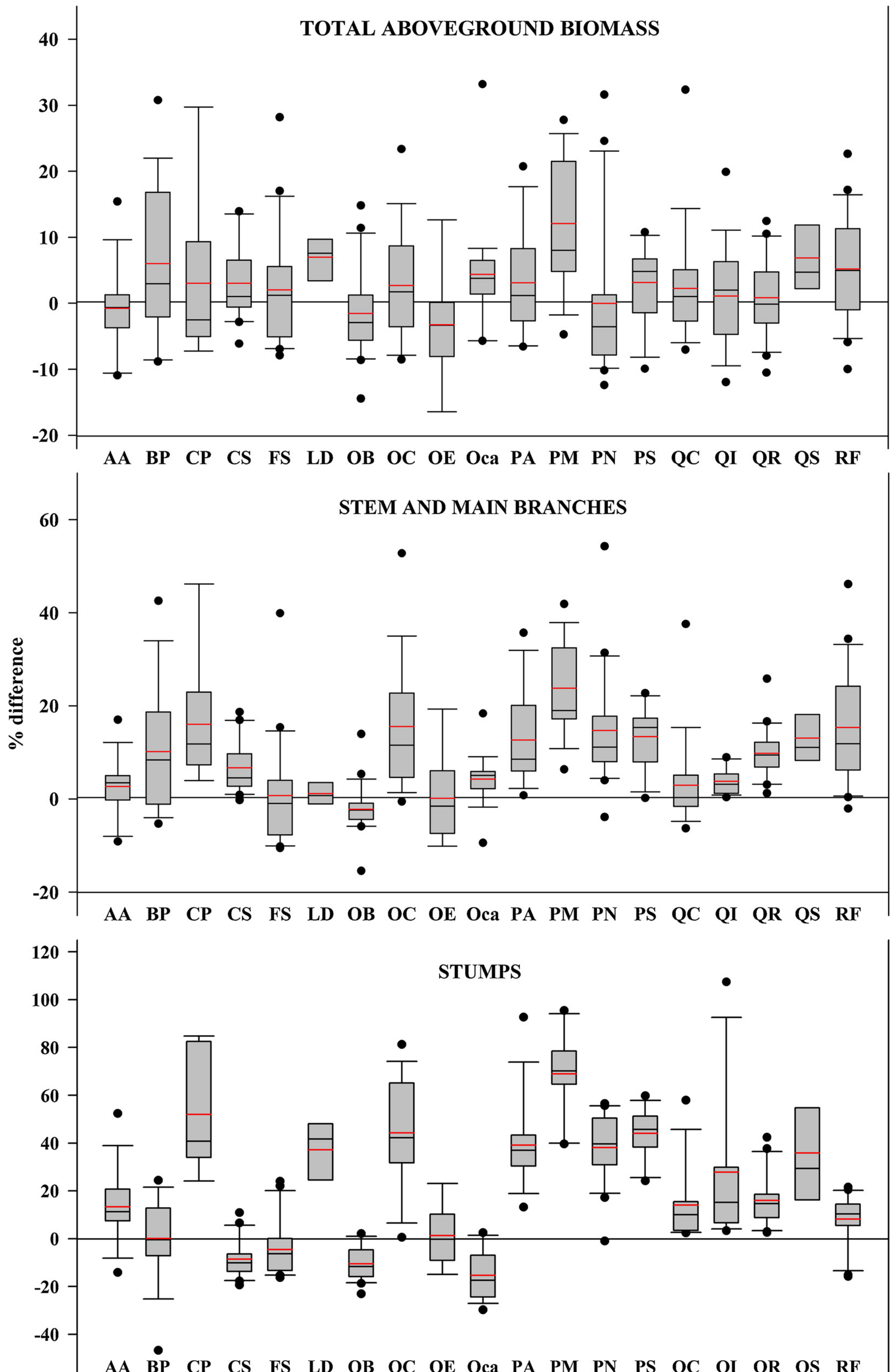

Fig. D.1. Boxplots of the percentage differences between the total aboveground biomass estimated through the selected volume to biomass equations and the biomass reported by the INFC for total aboveground biomass, stem and main branches, and stumps, for each forest type, based on the selected species. The mean percentage difference (red line, inside the box plot), median (black line inside the box plot), 25th and 75th percentile (boundaries of the box), 10th and 90th percentile (error bars) and outlying points are reported. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

Table E.1
Main parameters modified to calibrate the DOM pools, based on the default values reported by CBM model (Step 1), the values reported by Smyth et al. (2010) (Step 2) and a range of values reported by White et al. (2008) (Step 3). The following DOM pools were considered: very fast aboveground (VF Ab), very fast belowground (VF BG), fast aboveground (F AG), fast belowground (F BG), medium (Med), slow aboveground (S AG) and slow belowground (S BG). The first row reports the correspondence with different GPG pools (IPCC, 2003) as reported by Kurz et al. (2009). The final values used by model run were reported by Fig. 5.

GPG pools		Litter	Soil	Litter	Dead Wood	Litter	Soil	Litter
PARAM.	Step	VF AG	VF BG	F AG	F BG	Med.	S AG	S BG
Decay rate organic matter	1	0.355	0.5	0.1435	0.1435	0.0374	0.015	0.0033
	2	0.35	0.5	0.190	0.232	–	0.015	0.0033
	3	0.284–0.426	0.4–0.6	0.1–0.29	0.1–0.29	0.01–0.08	0.002–0.02	0.0008–0.004
q 10	1	2.65	2	2	2	2	2.65	1
	2	2.65	2	3.51	3.4	–	2.65	1
	3							
Prop. to atmosphere	1	0.815	0.83	0.83	0.83	0.83	1	1
	2	0.815	0.83	0.815	0.815	–	0.815	–
	3	0.742–0.888	0.55–0.85	0.7–0.9	0.7–0.9	0.7–0.9	–	–

In three cases there was a discrepancy between the original characteristics of the main species, as defined by our input dataset (i.e., the forest categories) and the best-matching FT. According to the original model design, HW and SW species defined broadleaved and conifers species, respectively (see Table D.1, last column). In our case, two SW species were selected for three broadleaved categories (i.e., Eastern white pine for Other broadleaves and Red pine for Hornbeam forests and Riparian forests). Since during the model run different litterfall rates are defined for softwoods and hardwoods species (i.e., 0.11 and 0.95, for SW and HW, respectively), the amount of biomass transferred from the living foliage biomass pool to the litter pool may be underestimated for these three categories. The same issue may also be considered for Larch, i.e., a deciduous conifer species that was associated to an HW species (Eastern white cedar).

Appendix E. Calibration of DOM parameters

To calibrate the DOM parameters, we compared the results provided by the model with data reported in the literature for some Italian regions. Since these studies (Petrella and Piazzi, 2005; Tonolli and Salvagni, 2007; Garlato et al., 2009) referred to data collected between 2001 and 2007, we compared them with the 2005 model output. Parameters defined for these regions were then applied to the entire dataset. The analysis was based on the following steps (see Table E.1 for a detailed description of the main parameters' range applied to calibrate DOM pools):

1. Application of the default parameters provided by the model.
2. Replacement of the default parameters (Version 1.2) applied in step 1, with the parameters from a 12-year study on forest litter and wood block decay in Canada (Smyth et al., 2010).
3. Parameters affecting litter and dead wood pools (i.e., turnover rate and snag fall rate) were further modified, according to the data ranges provided by a sensitivity analysis on the DOM–CBM sub-model (White et al., 2008).

For the DOM initialization phase (explained above and in Kurz et al., 2009), we set the average number of years between stand-replacing disturbances (i.e., fire as suggested by the default assumptions for the initialization of the model's DOM pools) to 250 years. A long interval was selected because stand-replacing natural disturbances such as fires, storms, landslides or avalanches (mainly for the alpine forests) are not very common for Italian forests. A clear cut with slash-burn was used as last pass disturbance event applied at the end of the iteration process.

According to Tabacchi et al. (2010), in 2005 natural mortality affected 3.73 Mm^3 of the aboveground biomass. Since the total aboveground growing stock reported by INFC is equal to 1269 Mm^3, we estimated an annual turnover rate equal to 0.0029 (i.e., about 0.3% per year). This value was applied to the stem pool, while for the branches we assumed an annual turnover rate equal to 0.02.

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