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FOREST CARBON SEQUESTRATION: THE IMPACT OF FOREST MANAGEMENT

1. INTRODUCTION

Regardless of their geographical location, forests play an important role in CO_2 fixation. Carbon stored in terrestrial ecosystems is distributed among three compartments: living plant biomass (stem, branches, foliage, roots), plant detritus (fallen branches and cones, forest litter, tree stumps, tree tops, logs) and soil (organic mineral humus, surface and deep mineral soil). Trees acquire energy for their living structures through photosynthesis, which requires CO_2 captured by stomata in the leaves. Part of the captured CO_2 is used to create living biomass, while the remainder is released back into the atmosphere through autotrophic respiration. When leaves or branches die and decompose, they increase soil carbon and also release a small amount into the atmosphere through heterotrophic respiration.

Recent climate changes have resulted in highly variable weather patterns and the general trend of rising temperatures is increasing evapotranspiration in forest ecosystems. A drop in available water for vegetation growth is expected to accompany this rise in temperature. Plants will respond to reduced water availability by closing stomata, resulting in lower rates of gas exchange. Although higher CO_2 concentrations from burning fossil fuels would be expected to increase photosynthetic rates, closed stomata may mitigate any positive impact in this regard. In areas where water is the limiting factor for growth and survival, inter- and intra-specific competition will be more acute, especially during the regeneration phase. Additionally, distribution of the energy captured by plants among different functions such as stem increment, branch and leaf formation, flowering, and fructification may be altered under the erratic weather conditions that climate change might generate.

Given that each tree species has an optimum temperature range for development, a widespread rise in temperature will modify the competitive balance between species and may alter species distribution patterns. Though adult trees are very unlikely to suffer generally from sudden death, this is possible in extreme conditions such as the drought affecting holm oaks (*Quercus ilex* L.). For example, Prieto-Recio *et al.* (2015) found that water deficits and high competition triggered the decline of *Pinus pinaster* Ait. stands in the Iberian Peninsula. Other

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problems related to regeneration and the initial development of forest species in certain areas will likely occur also. Ruano *et al.* (2009) identified water availability as a key factor in the germination and early growth of *Pinus pinaster* and later found water stress to be a mediator in seed production (Ruano *et al.,* 2015). Changes in precipitation amounts and regimes will likewise reduce natural regeneration in Mediterranean forests. Fujimori (2001) pointed out that these effects will be especially serious at the margins of plant distributions, where competition for resources is more pronounced. Disease and insect attacks may also become very severe (Melillo *et al.*, 1996). As a result, climate change has been classified as predisposing factor for forest decline (Hennon *et al.,* 2009). Conservation or even promoting greater biodiversity, with regards to both species and genotypes within each species, should constitute a management priority for reducing the effects of climate change. Mixed-species and other complex forests are potentially more adaptable to climate change, so promoting these forest types can help managers to cope with increasing uncertainty (Bolte *et al.,* 2009; Kolström *et al.,* 2011). Greater diversity of tree species can also limit damage from pests (Jactel and Brockerhoff, 2007) and diminish the risk of biological invasion, according to the associational resistance hypothesis (Barbosa *et al.,* 2009).

Helms (1998) defined silviculture as "the art and science of controlling the establishment, growth, composition, health and quality of forests and woodlands to meet the diverse needs and values of landowners and society on a sustainable basis". He subsequently defined forest management as the practical application of principles from a variety of disciplines, including biology, ecology, and economics, to the regeneration, density control, use, and conservation of forests. Silviculture and forest management were developed as sciences in Europe in the 18th century to satisfy the regular, continuous need for fuel and construction wood. Used together, they can mitigate the impact of climate change through four fundamental strategies: (1) conserving and maintaining carbon accumulated in the forests; (2) sequestering or incrementing the carbon retained in the forests and wood products; (3) replacing fossil fuels with biomass-derived fuels; and (4) reducing the use of products that require fossil fuels for manufacturing through use of renewable forest products such as wood, resin, and cork. Maintaining forest areas and using forests as a source of renewable energy will have the greatest impact worldwide.

In this chapter, we describe alternative ways in which forests and forestry can help to mitigate climate change, along with the potential impact of these activities. The three carbon storage compartments should be considered in all impact estimates. Carbon content in living biomass is easily estimated via species-specific equations or by applying factors to oven-dry biomass weights (e.g., Ibañez *et al.,* 2002, Herrero *et al.,* 2011, Castaño and Bravo, 2012). Litter carbon content has been analysed in many studies on primary forest productivity, though information regarding the influence of forest management on litter carbon content is less abundant (Blanco *et al.,* 2006). In the last decade, efforts have been made to assess soil carbon in forests, but studies on the effect of forest management on soils show discrepancies (Lindner and Karjalainen, 2007). Hoover (2011), for example, found no difference in forest floor carbon stocks among stands subjected to partial or complete harvest treatments in the United States.

2. FOREST MANAGEMENT AND CARBON SEQUESTRATION DURING THE LAST CENTURY

In its reports regarding on mitigation, the Intergovernmental Panel on Climate Change (IPCC, 2001, 2007 and 2014) warned of the temporality of forest carbon deposits and the possibility of extensive tree mortality from large-scale disturbances such as droughts and forest fires, leading to massive emissions. Globally, the more prominent carbon loss in tropical zones due to human-caused deforestation is being offset by expanding forest areas and increasing wood stocks in temperate and boreal forests.

Human activities and land use have historically affected carbon storage, emissions, and sequestration. For example, U.S. forests were carbon sources from 1700 to 1945. Since then, fire suppression and forest renewal in abandoned farmland have reversed the trend and forests have become carbon sinks (Houghton *et al.,* 1999). Woodbury *et al.* (2007) found that total U.S. carbon stocks had increased since 1990 and were expected to continue increasing through 2010. In contrast, Pacala *et al.* (2001) reported a stable carbon sink in the continental U.S. (excluding Alaska), with similar values for 1980-1989 and 1990-1994. China offers another example of how national forest management can alter carbon storage trends. After the social revolution in 1949, the carbon content in living biomass decreased due to human pressure on forest resources. From 1970 to 1998, afforestation and reforestation programs were implemented to increase forest land and, subsequently, stored carbon (Fang *et al.,* 2001). This effort continues and its impact on forest carbon stocks will be relevant in future decades (Zhou *et al.*, 2014). Bellassen and Luyssaert (2014) pointed out how assessments of forests as carbon sinks or sources rely on specific hypotheses about processes, including the carbon-neutral role of mature forests and the life-cycle of wood products. Clarifying the underlying processes involved in forest carbon stocks and cycles will reduce uncertainty and generate more reliable predictions, possibly confirming the persistence of forests as carbon sinks.

As noted earlier, the positive effects of forests as carbon sinks are endangered by large-scale disturbances. Liu *et al.,* (2002) showed that the forests of Ontario, Canada could have been considered carbon sinks between 1920 and 1975, but became carbon sources after that because of large-scale natural and human disturbances including wildfires, pest infestations, and extensive harvesting. Schmid *et al.*, (2006) noted that forests with minimal management serve as carbon sinks in the short term, but accumulated biomass will increase risks of fire and pest occurrences.

In many European countries, forest management did not begin until the 19th and 20th centuries. Once implemented, forest management planning, activities, decisions, and results have since been recorded. Consequently, we can determine how CO_2 fixation as forest biomass has evolved. Current forest management activities are improving silvicultural activities aimed at increasing the quantity of fixed carbon in the forests. For example, (Montero *et al* (2004) found that from 1993 to 2003 net carbon fixation increased by 6.28% in the woodlands of Monte de Valsaín (Segovia, Central Spain) Similarly, in the Pinar Llano woodlands of Valladolid (Northern Plateau, Spain) the amount of fixed carbon was expected to increase by 7.23% in the next 10 years, based on the rate of gain presented in the Management Project Review (Martín, 2005). Bravo *et al.* (2007) analysed different forest areas in Spain and found that the annual CO_2 sequestration rates in temperate and Mediterranean forests ranged between 0.95 and 4.96 % (Table 1). These results were obtained by comparing outcomes from the second and third Spanish National Forest Inventories (INF2, INF3) using biomass equations by Montero *et al.*, (2005) and converting to equivalent carbon.

Table 1. Annual CO_2 sequestration rates in temperate and Mediterranean forests in Spain (adapted from Bravo *et al.*, 2007). IFN2 and IFN3 are the second and third Spanish National Forest Inventories, respectively.

Zone	Years	IFN2 (10^3 Tn)	IFN3 (10^3 Tn)	Annual rate (%)
Cantabrian range	1991-2000	41696	45433	0.9583
Castillian plateau	1992-2992	17623	21635	2.0723
Basque Mountains	1994-2003	32608	50601	5.0035
Catalonian coastal range	1991-2001	15591	19532	2.2792
Demanda range	1992-2003	5195	95244	5.6200
Pyrenees	1994-2003	120159	185812	4.9628
Central range	1991-2002	57164	72404	3.2069
Toledo Mountains	1992-2001	9619	11802	2.2986
Sierra Morena	1994-2001	15518	19542	2.3324

Bravo *et al.*, (2010) studied the evolution of fixed carbon dioxide in the Mediterranean maritime pine forests (*Pinus pinaster* Ait.) of Almazán (Soria, Central Spain). They analysed a century of woodland management in Almazán, from 1899 to 1999, obtaining the numbers of trees by size from the original planning documents and successive planning revisions. This information was combined with the biomass equations developed by Montero *et al.*, (2005) to reconstruct the evolution of CO_2 fixation in these forests. Carbon sequestration increased during the century studied, oscillating between 0.78 and 3.11 Tn/ha per year (Fig. 1). The only exception was the period immediately after to the Spanish Civil War (1936-9), when greater pressure on natural resources due to poverty caused a decrease in biomass (and a reduction of 1.49 Tn of CO_2/ha per year). The CO_2 forest biomass sequestration levels in these pinewoods did not recover until

15 years after the end of the war. Forest management under a sustained yield paradigm that also maintains or increases other forest values and services has been traditionally implemented in the Almazán forests. This management regime maintains or increases forest carbon stocks over the long term, while producing goods and values according to IPCC goals (2007 and 2014). The increases in carbon sequestration achieved in the Almazán forest align with evidence by Nabuurs *et al.*, (2003), which showed that European forests increasingly became carbon sinks between 1950 and 1999. Their carbon calculations were based on simulations of carbon storage using net biomass production (i.e., final production including harvest and disturbances). Later, Nabuurs *et al.*, (2007) stated that sustainable forest management strategies designed to increase or maintain forest carbon stocks while producing a constant annual yield of goods (wood, fibre, etc.) and environmental services (water, biodiversity, etc.) will generate the largest long-term sustained mitigation benefit.

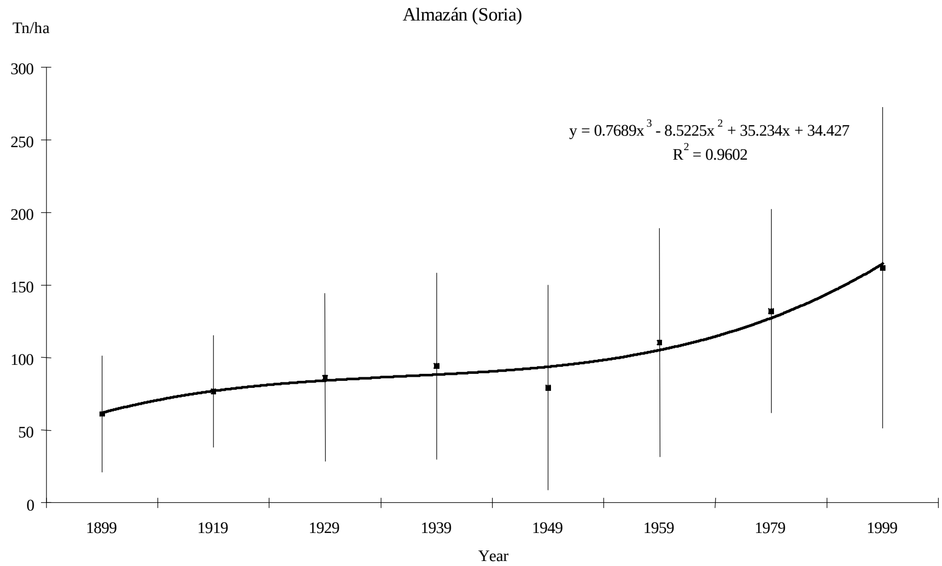

Fig. 1. Evolution of CO_2 sequestration in the pine woods (Pinus pinaster Ait.) of Almazán (Soria, Northern Spain) during the 20th century

3. STRATEGIES TO IMPROVE CARBON SEQUESTRATION

Carbon storage in forests and forest products has been proposed as an appropriate strategy for mitigating the effects of climate change, In spite of this, forest products were excluded from the Kyoto protocol. To a certain extent, carbon storage in forests buys time, until we find more definitive solutions to our dependency on energy from fossil fuel. However, forests can easily become carbon sources rather than carbon sinks (Kurz and Apps, 1999, Gracia *et al.,* 2001, Reichstein *et al.,* 2002). Changes in the regimes of natural disturbances such as fire, pests or drought (Fuhrer *et al.,* 2006, Sohngen *et al.,* 2005, Ciais *et al.,* 2005), can affect major forest functions, forestry outputs and forest stability. Metsaranta *et al.* (2010) reported that Canadian forests will likely be carbon sources until 2030, and become carbon sinks after 2050, based on simulations of the impact of future fire and insect disturbances.

Biomass and carbon accumulation in forest stands can be increased through a variety of management options (Gracia *et al.,* 2005). These include fire, disease, and pest control; increasing rotation lengths (i.e., time to harvest); density regulation; fertilization and other activities to improve soil nutrients; species and genotype selection; management of post-harvest residues and advances in fibre processing and biotechnology. Such activities could increase the carbon accumulation rate by 0.3 to 0.7 Mg C ha^{-1} yr^{-1} (Gracia *et al.*, 2005). Management practices that alter species composition, rotation lengths, and thinning regimes, or that result in forest conservation, increased forest land, and soil conservation can also increase carbon sequestration in forests.

3.1. Species composition

Carbon storage varies according to species composition and site quality (Bravo *et al.,* 2008). For example, the amount of carbon per unit biomass is greater in conifers than in broadleaf trees (Ibáñez *et al.,* 2002). In Mediterranean areas, Scots pine (*Pinus sylvestris*) stands store more CO_2 than pure oak (*Quercus pyrenainca*) stands (Bogino *et al.,* 2006), while mixed oak-pine stands store intermediate levels. Differences among pine species have also been reported (Bravo *et al.*, 2008).

Many studies have reported greater forest productivity with increased species diversity (e.g. Vilà *et al.,* 2007; Paquette and Messier 2011). As a result, mixed-species forests are likely to have higher carbon storage capacity. Similarly, the productivity by space occupancy of many species is frequently higher when admixed with other species due to facilitation and/or complementarity, often resulting in over-yielding or even transgressive over-yielding compared to corresponding monospecific stands (e.g. Río and Sterba, 2009; Condés *et al.,* 2013; Pretzsch *et al.,* 2015). Forest productivity is generally evaluated based on volume or aboveground biomass per unit area, calculated via tree volume or tree biomass allometric equations. These were often developed using sample data from monospecific stands, resulting in less accurate biomass estimates (Forrester and Pretzsch, 2015). Carbon content per unit of biomass may also vary, depending on whether stands are mixed or monospecific. Additionally, belowground biomass is difficult to estimate and often excluded from estimates, and niche complementarity between species can also occur belowground (Brassard *et al.*, 2013). These issues underscore the need for more research focusing on carbon storage in mixed-species forests, in order to determine whether they fix more CO_2 than monospecific forests.

Species composition can also modify soil carbon storage, as tree species identity mediates soil carbon distribution between the forest floor and mineral soil (Vesterdal *et al.* 2013). There are indications of a continuum species effect mediated by local conditions rather than a global dichotomy effect such as deciduous vs. conifer (Prescott and Vesterdal, 2013). In a study of beech (*Fagus sylvatica* L.) forests and *Pinus nigra laricio* plantations in Calabria (Southern Italy), Scarascia-Mugnozza *et al.*, (2001) found that beech stands stored 1.47 times more soil carbon than pine plantations. Co-existence with species that fix nitrogen can also increase carbon accumulation. Chiti *et al.*, (2003) found that mixed oak-alder (*Quercus robur* L.-*Alnus cordata* Desf.) stands stored 1.18 times more carbon in the soil than pure oak stands in Tuscany (Italy), probably due to a higher humification rate. The mix of tree species affects soil fauna assemblages (Chauvat *et al.*, 2011), whereas litter type and initial leaf-litter concentrations of co-existing species affect soil processes such as litter decomposition and nutrient release (Aponte *et al.*, 2013).

The selection of the best species composition for a stand depends on many factors (management objectives, site characteristics, etc.), and CO_2 storage should also be included as an objective. Silvicultural treatments can be used to alter the species composition of stands, mainly through selection of species for regeneration, and through manipulation of species composition through thinning and other stand-tending treatments in established plantations and forests.

3.2. Rotation length

During immature and mature stages of stand development, forests are carbon sinks. In older forests, carbon sequestration may continue to increase slowly or may decrease slightly. Rotation length can be extended in order to maximize or maintain forest carbon sinks while obtaining other goods and services. Different criteria can be used to determine the appropriate rotation for obtaining forest products while achieving forest sustainability. One widely-used criterion in the management of regulated forests is to set the rotation length at biological rotation, or maximum mean annual increment (MAI) of volume per unit area. This practice maximizes longer-term wood production (several rotations), while promoting other products and services that society demands (wild mushrooms, hunting, ecosystem conservation, etc.). Rotation has a two-fold impact on carbon storage in forests (Table 2). Under rotations longer than the biological rotation, the proportion of carbon in the final harvest relative to intermediate harvests is greater (Bravo *et al.*, 2008). Similarly, since products from harvests are often destined for long-term uses (e.g., furniture, construction etc.), products made from harvests after longer rotations result in greater carbon storage.

*Table 2. The impact of species, site quality and rotation on carbon sequestration in stands of Scots pine (*Pinus sylvestris *L.) and Mediterranean Maritime pine (*Pinus pinaster *Ait.) (adapted from Bravo et al., 2008).*

Species	Site index*	Rotation (years)	Mean annual carbon growth (MAI) (t year $^{-1}$)	% carbon final harvest
Pinus sylvestris L.	17	83	2.16	54.61
		137	1.47	59.60
	23	69	2.99	68.12
		122	2.42	77.66
Pinus pinaster Ait.	15	101	1.28	75.19
		149	1.06	79.72
	21	83	1.89	71.91
		128	1.57	78.06

* Site index is the dominant height in m at 100 years (*Pinus sylvestris*) or 80 years (*Pinus pinaster*)

If the rotation length is very long, tree mortality rates will increase, resulting in an increase in structural diversity with dead and fallen trees. This, along with regeneration in the gaps created is related to an overall increase in species diversity (Franklin *et al.*, 1997). Tree mortality contributes to coarse woody debris (CWD), a key carbon reservoir in mature and old-growth forests. Differences in CWD stocks have been found among forest types (Herrero *et al.*, 2010 and 2014a). The decomposition rates of dead woody materials varies with the species, size, type of material (i.e., bark, sapwood or heartwood), and site conditions (i.e., temperature, humidity, etc.). Dead wood has an important impact on carbon storage in forest systems because it may increase the risk of perturbations, resulting in sudden outbreaks of fires, pests, and pathogens. Other impacts on the amount of soil carbon may also occur due to increased rotation lengths. Using the CO2FIX model, Kaipainen *et al.* (2004) reported a decrease in soil carbon stocks for some cases in Europe when the rotation length was increased.

There is evidence that biomass allocation among different components varies with tree age. For *Pinus sylvestris* and *Pinus pinaster*, Bravo *et al.,* (2008) found that with age biomass allocation to stems increases, while allocation to branches decreases. Added to the importance of larger stems for carbon storage in stands, as well as in wood products created from these stems, the biomass distribution in trees harvested after longer rotations had a considerable impact on the possible use of harvest debris to generate energy. Bravo *et al.* (2008) have demonstrated that the percentage of biomass for pinewood branches between 2 and 7 cm in diameter decreases with age.

The proportion of carbon stock in the final harvest relative to total fixed carbon in the stand is higher for longer rotations. However, a shorter rotation is associated with higher carbon MAI values at rotation, regardless of the site index (Bravo *et al.*, 2008, Table 2). Longer rotations on poor sites can result in carbon storage similar to that of shorter rotation on good sites, as shown for *P. sylvestris*. Additionally, a longer rotation on a poor site may produce stems large enough to be suitable for lumber and other products. Thus, long rotations could be applied to the poorest sites in order to achieve both carbon sequestration and timber value objectives. Bravo and Diaz-Balteiro (2004) showed that more extensive management systems that involve lengthening rotations result in a loss of economic return compared to traditional management with shorter rotations. However, when carbon sequestration income is included in the analysis, longer rotation alternatives present a positive land expectation value. Increasing harvest rotation length would lead to reduced harvest rate over a landscape. Under such circumstances, some carbon pools will increase (e.g., carbon in standing trees) while others decrease (e.g., carbon in wood products) (Kurz *et al.*, 1998). Therefore, the carbon pool dynamics on a broad temporal and spatial scale should be included in management planning.

3.3. Thinning

Management of tree density by thinning is one of the most important silvicultural interventions for achieving both economic and ecological objectives (Río, 1999):

- To reduce competition in order to procure biological stability and improve health.
- To regulate or maintain the specific composition and to prepare the stands for natural regeneration.
- To obtain production yield at early stages, in such a way as to maximize production at the end of the rotation.
- To increase the value and dimensions of the remaining products.

However, thinning will affect the amount of stored carbon. In particular, aboveground tree biomass is reduced immediately after thinning, along with litterfall inputs and accumulation on the forest floor. However, carbon sequestration rates may increase after thinning, as the growth rates of residual trees are altered.

Long-term experiments are essential for obtaining knowledge about the effects of thinning on carbon storage and sequestration rates. A few studies have shown that unthinned stands present higher carbon stocks in tree living biomass than thinned stands, because higher stocking from moderate or heavy thinning results in lower carbon stocks over the long-term (Skovsgaard *et al.*, 2006; Powers *et al.*, 2011; Ruiz-Peinado *et al.*, 2013 and 2014). However, in some cases very light thinning has a positive effect on carbon storage, as Keyser and Zarnoch (2012) found.

Though maintaining a high tree density could maximize on-site carbon stock, it may also increase the risk of natural disturbances (Jandl *et al.,* 2007). Increasing off-site carbon storage via thinning may prove a better strategy, especially in high risk areas. Tree carbon removed by thinning operations should be included in calculations of total carbon stocks and carbon sequestration rates in order to compare thinning intensities, as is often done for volume production (Assmann, 1970).

Along with thinning intensity, the type of thinning may also affect carbon stocks and sequestration rates. Hoover and Stout (2007) showed that thinning from below presented higher carbon stocks in tree biomass than thinning from the middle or from above. With thinning from below, growth is concentrated in the larger trees that are retained: smaller trees with lower net productivity are removed, making resources available to the residual trees. Similarly, D'Amato *et al.,* (2011) found equal amounts of carbon in tree biomass with thinning from below as with a combination of thinning from above and from below (i.e., multiple thinning events) over the long term.

Thinning can also reduce the amount of deadwood, since dying or dead trees are often removed in thinning operations. Other carbon pools such as soil carbon (forest floor and mineral soil) may also be affected. Reduced density from thinning may alter microclimatic soil conditions, thereby affecting soil temperature and moisture. Thinning activities may also result in soil compaction as well as mixing of forest floor litter with upper soil layers. As noted earlier, fewer trees imply lower litterfall inputs and higher carbon losses as a consequence of higher respiration rates (Jonard *et al.,* 2006).

The harvesting method can result in different impacts of thinning on forest carbon. Whole-tree harvesting might have more intense impact than stem-only harvesting, where thinning residues that could reduce soil impacts such as soil compaction are retained on-site (Tarpey *et al.,* 2008; Han *et al.,* 2009). In a meta-analysis, Johnson and Curtis (2001) found that harvesting had no statistically significant effect on soil carbon stock. However, these authors also reported differences depending on the harvest method used: there was a slight reduction in soil carbon stocks when whole-tree harvesting was applied and a moderate increase with sawlog harvesting; but this was restricted to coniferous species. The Nave *et al.,* (2010) meta-analysis found that harvesting reduced soil carbon stocks in a small but significant way: forest floor carbon stocks decreased markedly and no influence was detected in the mineral soil, though great variation was identified between soil orders. This forest floor carbon stock reduction tendency for thinned stands is corroborated by several studies (e.g., Vesterdal *et al.,* 1995; Jonard *et al.,* 2006; Powers *et al.,* 2012; Ruiz-Peinado *et al.,* 2013), though other authors have reported little or no influence (Novák and Slodicák, 2004; Chatterjee *et al.,* 2009; Jurgensen *et al.,* 2012).

Analysis of the effects of thinning on total ecosystem carbon should include all pools: tree biomass (above and belowground), understory (shrub, herbaceous…), deadwood, forest floor and mineral soil. To obtain the most complete results, the carbon removed in thinnings should be also incorporated into the analysis, as mentioned earlier.

The works of Ruiz-Peinado *et al.*, (2013) and Bravo-Oviedo *et al.*, (2015) in Spain indicated no significant influence of thinning on total ecosystem carbon stock, when compared to unthinned stands at the end of the rotation period. In another study (Ruiz-Peinado *et al.*, 2014), early thinning in the middle of the rotation period slightly reduced the total carbon stock .

In a chronosequence study on Pinus resinosa, Powers *et al.*, (2012) found that thinning did not have a significant effect on total ecosystem carbon stock. However, in a previous study of the same species, Powers *et al.*, (2011) found that thinning reduced the total carbon stock when these treatments were applied at 5-10 year interval. These results suggest that thinning rotation period is another important aspect of forest management to consider in relation to carbon storage.

Stand age also influences the carbon amounts that are stored in the different forest compartments. Stands have different carbon sequestration rates at different ages, even when managed under a similar thinning plan. Using data from the studies of Ruiz-Peinado *et al.*, (2014) and Bravo-Oviedo *et al.*, (2015) on *Pinus sylvestris*, Figure 2 shows that the greatest carbon pool was found in the aboveground tree biomass of medium-young stands, whereas in old stands (near the rotation period for this species) the most important pool was located in the mineral soil. Carbon removed by thinning constitutes an important pool in both types of managed stands, especially if we consider that wood products can store carbon for long periods.

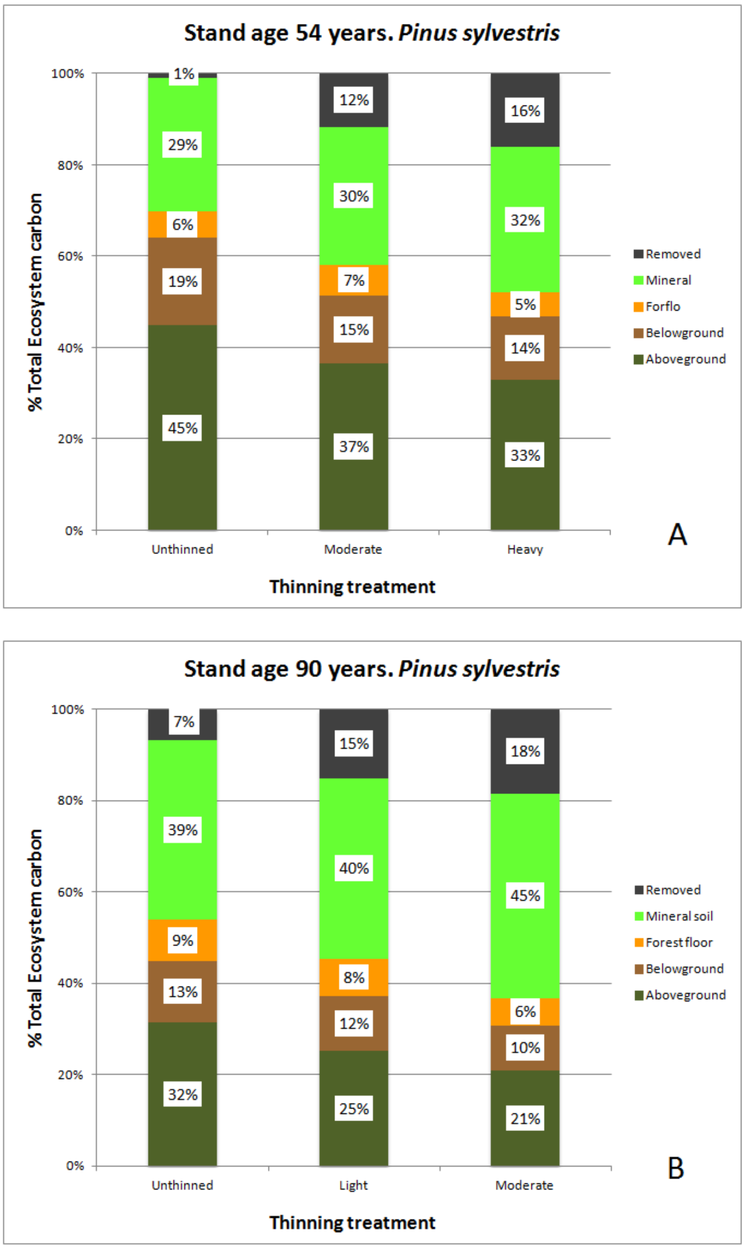

Figure 2. Total carbon proportions found in the different pools of *Pinus sylvestris* stands after 30 years of forest management and 3 thinnings. A: data obtained from a 54 year-old afforested stand (Ruiz-Peinado *et al.*, 2014). B: data obtained from a 90 year-old natural stand (Bravo-Oviedo *et al.*, 2015).

Pre-commercial thinning should be applied to very dense stands, (e.g. in natural regeneration after a forest fire), in order to reduce tree density to near the suggested values. Results from a study of *Pinus halepensis* by De las Heras *et al.*, (2013) showed that applications of early and heavy treatments presented carbon amounts similar to those of unthinned plots. However, thinning accelerated cone production and stand maturity (2-5 years after) and tree growth (2-4 years after), thereby increasing stand resilience against perturbations such as forest fires (Ruano *et al.*, 2013). Similarly, Jiménez *et al.*, (2011) found for *P. pinaster* that early and moderate thinnings resulted in no total carbon biomass differences compared to unthinned plots.

In general, the findings and results of the different studies indicate that the effect of thinning on soil carbon stocks is not quite significant, although a high degree of variation exists according to species, harvesting methods, soil types, etc. Living biomass decreased with the thinning interventions in the long-term (reduction of tree density). However, in areas with high risk of fire or other disturbances (winds, heavy snow, etc.) or when pests or disease may endanger the ecosystem, moderate or heavy thinning helps maintain tree cover and improve carbon sequestration on-site as well as off-site, in wood products or as bioenergy.

3.4. Conservation of forests

Although forestry activities have different carbon mitigation potentials depending upon ecosystem features, the short-term carbon mitigation benefits of conserving current forests by reducing deforestation outweigh the benefits of increasing forest area (IPCC, 2007). Between 1990 and 2005, 13 million ha per year of forest land were lost to other uses (FAO, 2006), and deforestation rates were highest in South America, South and Southeast Asia and Africa (Table 3). Forest loss rates are currently decreasing worldwide (from 7.3 million ha per year in 1990 to 3.3 million ha per year in 2015), but total forest area declined by 3% between 1990 and 2015) (Keenan *et al.*, 2015), with differences between climatic domains. While subtropical and boreal forests remain stable, tropical forest area decreased and temperate forest area increased (Keenan *et al.*, 2015) Between 2000 and 2005, Brazil (3 million ha per year), Indonesia (1.8 million ha per year) and Sudan (0.6 million ha per year) suffered the largest amount of deforestation (FAO, 2006). Though forest area loss was controlled between 2010 and 2015 (Keenan *et al.*, 2015), Brazil still presents the highest forest area loss values (984 thousand ha per year), followed by Indonesia (684 thousand ha per year), Myanmar (546 thousand ha per year) and Nigeria (410 thousand ha per year). Conserving forests by reducing deforestation and degradation, especially in threatened areas such as the tropics, is the most effective short-term strategy for carbon stock preservation and may be included in the United Nations Framework Convention on Climate Change (UNFCC) as the official mechanism of the Climate Change Agreement. Effective forest protection will facilitate carbon sequestration while adaptive management of protected areas will result in biodiversity conservation and reduced vulnerability to climate change (Nabuurs *et al.*, 2007).

Table 3. Forest area by regions between 1990 and 2015 (adapted from Keenan et al., 2015 based on FAO, 1995, 2006, 2010 and 2015)

	Forest area (1000 ha)				
Region	1990	2000	2005	2010	2015
Africa					
East and South Africa	319785	300273	291712	282519	274886
North Africa	39374	37692	37221	37055	36217
West and Central Africa	346581	332407	325746	318708	313000
Asia					
East Asia	209198	226815	241841	250504	257047
South and Southeast Asia	319615	298645	296600	295958	292804
West and Central Asia	39309	40452	42427	42944	43511
Europe	994271	1002302	1004147	1013572	1015482
America					
Caribbean	5017	5913	6341	6745	7195
Central America	26995	23448	22193	21010	20250
North America	720487	719197	719419	722523	723207
South America	930814	890817	868611	852133	842011
Oceania	176825	177641	176485	172002	173524
Global	**4128271**	**4100602**	**4032743**	**4015673**	**3999134**

3.5. Increasing forest area

In recent decades, reforestation of marginal lands in temperate zones has increased through natural and artificial reforestation of abandoned farmland. Increased forest area in Europe in the last quarter of the twentieth century, prior to similar trends in the US, has led to increased carbon reserves in living biomass as well as in the soil (Liski *et al.*, 2002). Between 2000 and 2005, Mediterranean countries (Spain, Portugal, Italy and Greece), Vietnam and China were the greatest contributors to increases in forest area in the world, while tropical countries were the greatest contributors to decreased forest area. China (1.98 million ha per year), Spain (0.39 million ha per year) and Vietnam (0.42 million ha per year) dramatically increased their forest areas between 1990 and 2000 (FAO, 2006). From 2000 to 2010 Vietnam and Spain slightly reduce its forest area change (0.21 and 0.12 million ha per year respectively) but China increase its change in forest area (2.99 million ha per year) China still presented the highest rate of forest expansion (1.5 million ha per year between 2010 and 2015), though less than that of previous years (Keenan *et al.*, 2015)

Living biomass carbon stocks decreased steadily from 1990 to 2005, but by 2010 had recovered to year 2000 levels (Table 4) (FAO, 2006 and 2010). In 2010, living biomass carbon stocks increased (compared to 1990 levels) in Europe, East Asia and South America, but decreased in North Africa, West and Central Africa, South and Southeast Asia, Central America and Oceania. Finally, biomass carbon stocks remained fairly constant in West and Central Asia, East and South Africa, West and Central Asia and North America. Forests of South America and Africa constitute the largest carbon reservoirs; therefore, conservation of forests in these continents is crucial to mitigating climate change through forest management initiatives.

Forest plantations can be used in the Kyoto protocol for emission reduction accounting, but only in regulated circumstances and by some developed countries. Increased plantation area is the main forest activity that be used to counteract carbon emissions from fossil fuels in developed nations. Between 1990 and 2005, productive plantation area increased from 76.8 million ha to 109.3 million ha (Table 5) (FAO, 2006). China (26 % of global productive plantation area), United States (16 %), Russia (11 %) and Brazil (5%) are the leading countries. In that period, China increased plantation area by a factor of 1.665, or 759 thousand ha per year. China, Russia and United States together represented 71 % of new productive plantations between 1990 and 2005 (FAO, 2006). Protective plantations for conservation purposes increased from 20.4 million ha in 1990 to 30.1 million ha in 2005 (Table 5). Japan (35 %) and Russia (17 %) had the most area planted for protective purposes (FAO, 2006).

Table 4. Forest biomass carbon by regions between 1990 and 2010 (adapted from FAO, 2006 and 2010)

Region	Biomass carbon (Gigatons)			
	1990	2000	2005	2010
Africa				
East and South Africa	15.9	14.8	14.4	15.8
North Africa	3.8	3.5	3.4	1.7
West and Central Africa	46.0	43.9	43.1	38.3
Asia				
East Asia	7.2	8.4	9.1	8.8
South and Southeast Asia	32.3	25.5	21.8	25.2
West and Central Asia	1.6	1.7	1.7	1.7
Europe	42.0	43.1	43.9	45.0
America				
Caribbean	0.4	0.5	0.6	0.5
Central America	3.4	2.9	2.7	1.8
North America	37.2	38.5	39.2	37.3
South America	97.7	94.2	91.5	102.2
Oceania	11.6	11.4	11.4	10.5
Global	**299.2**	**288.6**	**282.7**	**288.8**

The effect of plantations on carbon sequestration varies according to plantation type, objectives and management, including whether the plantation is primarily intended as a productive or conservation area. Protective plantations managed for conservation (through long rotation, for example) have a limited impact on carbon sequestration –since carbon sequestration rates decline in very old plantations– but can work as effectively on poor sites as short rotation does on better sites (Bravo *et al.*, 2008). In that sense, plantations for production (e.g., for wood biomass, wood for building material, etc.) represent a better carbon mitigation strategy (Lindner and Karjalainen, 2007). At each rotation, substitution with younger trees results in a net carbon emission mitigation effect.

Table 5. Forest plantation area by regions between 1990 and 2005 (adapted from FAO, 2006)

	Productive plantations (1000 ha)			Protective plantations (1000 ha)		
Region	1990	2000	2005	1990	2000	2005
Africa						
East and South Africa	2544	2712	2792	66	66	66
North Africa	6404	6158	6033	1840	2021	2192
West and Central Africa	1099	1453	1853	70	87	112
Asia						
East Asia	17909	23028	30006	11622	12490	13160
South and Southeast Asia	8896	10750	11825	3869	4451	4809
West and Central Asia	2120	2428	2583	2175	2518	2505
Europe	16643	19818	21467	4569	5574	6027
America						
Caribe	239	243	280	155	151	170
Central America	51	183	240	32	29	34
North America	10305	16285	17133	-	1047	986
South America	8221	10547	11326	10	27	31
Oceania	2447	3456	3812	1	3	21
Global	**76826**	**97061**	**109352**	**24408**	**28464**	**30114**

Starting 2010 Forest Resources Assessment (FRA), no information is provided regarding classification of forest plantation areas as productive or protective (FAO, 2010). The concept of planted forests (originated by afforestation or reforestation) is introduced in Table 6. China (1,932 thousand ha per year), United States (805 thousand ha per year), Canada (385 thousand ha per year) and India (251 thousand ha per year) accounted for more than 78.64 % of the new planted forest area between 1990 and 2010 (FAO, 2010).

Table 6. Forest area by regions between 1990 and 2010 (FAO, 2010)

	Planted forests area (1000 ha)			
Region	1990	2000	2005	2010
Africa				
East and South Africa	3500	3689	3813	4116
North Africa	6794	7315	7692	8091
West and Central Africa	1369	1953	2526	3203
Asia				
East Asia	55049	67494	80308	90232
South and Southeast Asia	16531	19736	23364	25552
West and Central Asia	4678	5698	5998	6991
Europe	59046	65312	68502	69318
America				
Caribbean	391	394	445	548
Central America	445	428	474	584
North America	19645	29438	34867	37529
South America	8276	10058	11123	13821
Oceania	2583	3323	3851	4101
Global	**178307**	**214839**	**242965**	**264084**

When forest plantation projects are intended to compensate for CO_2 emissions, baseline carbon fixation (situation prior to plantation) must be compared to the expected carbon fixation from the plantation. Also, a reliable monitoring and accounting program should be developed for land within the project boundaries. Carbon monitoring and accounting programs require a large database and fitted biometric models, including volume equations, biomass expansion factors, root-shoot ratios, and other previously fitted models. In some cases, prior models or data are not locally available and substitutions must be made. Guidelines have been developed to indicate the order of priority for use of substitutions: (1) existing local and species-specific models; (2) national and species-specific models; (3) species-specific models from neighbouring countries with similar ecological conditions; or, finally (4) global species-specific models, such as those from IPCC. Uncertainties arising from these biometric models and from sampling have to be considered in accounting (Temesgen *et al.,* 2015, Weiskittel *et al.,* 2015); limitations that affect biomass equations include (1) high variation in sampling areas and stands, (2) data gathering in limited areas, (3) methods that rarely include belowground biomass, (4) use of simple models that do not take into consideration autocorrelation problems or mensuration of key variables and (5) loss specific estimates due to grouping of species to fit robust models. In some cases these problems are solved by including belowground biomass in the equations (Herrero *et al.* 2014b), including crown size (Goodman *et al.,* 2014) as an independent variable in the models or using light detection and ranging (LiDAR) technology estimates of independent variables (Uzquiano *et al.,* 2014). Although the best solution for biomass estimation is difficult to establish, different alternatives should be explored that improve estimations (Weiskittel *et al.*, 2015) and that incorporate: (1) consistency in biomass data gathering across large geographical scales, (2) generation of open datasets compiling volume, biomass, carbon and wood density data, (3) use of such open datasets to evaluate and compare biomass models, (4) testing of new model forms using data from technology such as airborne and terrestrial LiDAR and (5) the application of appropriate available mathematical and statistical methods. Another source of uncertainty stems from the use of general default values for forest species when specific differences in gravity and carbon contents are widely-known (Herrero *et al.*, 2011, Castaño and Bravo, 2012) Specific differences can even occur between tissues for each species, as Herrero *et al.* (2011) determined for three Mediterranean pines (*Pinus nigra* Arn., *Pinus pinaster* Ait. and *Pinus sylvestris* L.) and Castaño and Bravo (2012) reported for two European oaks (*Quercus petraea* and *Quercus pyrenaica* Wild). Also, CO_2 losses due to plantation activities, such as burning of fossils fuels by machinery and biomass losses in site preparation prior to planting, have to be subtracted from the amount of carbon fixed. Different protocols have been approved for different plantation types and geographical areas (e.g., “Methodologies to Use Forestry as Mechanisms of Clean Devolopment, cases AR-AM0001 and AR-AM0003”[1]), which must be followed in order to obtain carbon credits from forest plantation activities.

A major economic limitation to plantations as a mitigation option is the high initial investment to establish new stands coupled with the delay (usually several decades) in generating revenue (Nabuurs *et al.*, 2007). Where forest expansion leads to a reduction of agricultural land area that in turn results in intensive farming practices, the conversion of mature forests to croplands or increased agricultural imports (McCarl and Schneider, 2001), will generate more emissions than potential sinks from plantations will occur globally.

3.6. Soil Conservation

Soils are the main terrestrial carbon sink. By conserving soil carbon, we can reduce CO_2 emissions and contribute to climate change mitigation. According to The Royal Society (2001), carbon stored in soils is three times the carbon stored in living biomass (1750 versus 550 PgC). Forests store around 50% of total soil carbon while representing only 30.3% of emergent lands (FAO, 2006). Soils contain the largest carbon stock in terrestrial ecosystems, representing 50.62% of total carbon in tropical forests, 62.75% in temperate forests and 84.31% in boreal forests (Fujimori, 2001).

[1] “Revised Approved Afforestation and Reforestation Baseline Methodology Case AR-AM00010 Facilitating Reforestation for Guangxi Watershed Management in Pearl River Basin, China” and “Case ARNM0018, Assisting natural regeneration on degraded land in Albania” , http://cdm.unfccc.int/methodologies/ARmethodologies/approved_ar.html.

The soil carbon pool and associated dynamic processes have not been studied to the same extent globally as carbon in living or dead biomass. Given the critical role of soils in overall carbon storage, conservation measures –including fire prevention and control– must be developed and implemented to conserve carbon pools in soil. Land reclamation via forest plantation on degraded land is another important carbon pool enhancement measure; plantations on formerly eroded soils can store up to 77% more carbon (Tesfaye *et al.,* 2016). Forest harvesting operations commonly result in short-term carbon losses from the soil (Turner and Lambert, 2000). In fact, research has indicated changes in soil carbon related to management intensity (i.e. removal or maintenance of slash, soil compaction or increased radiation due to open canopies), though these changes would not be significant over the longer term (Henderson *et al.,* 1995, cited in Paul *et al.,* 2002). Adequate management of harvest debris, which contains 20-35% of total tree carbon content, is crucial for maintaining soil carbon levels.

Although carbon pools in old-growth forests are considered to be in a steady state, Zhou *et al.,* (2006) showed that from 1973 to 2003, soil organic carbon increased at an average rate of 0.035% each year in old-growth stands (over 400 years old) in China. These results suggest that a longer rotation length may increase soil carbon, even though living biomass accumulation may have reached an asymptote. Soil carbon maintenance was found to be compatible with sustainable forest thinning practices that did not affect soil C stock in natural and planted Scots pine (Ruiz-Peinado *et al.,* 2014; Bravo-Oviedo *et al.* 2015) and afforested *P. pinaster* (Ruiz-Peinado *et al.,* 2012) in south-west Europe.

Regarding afforested areas, Paul *et al.,* (2002) reviewed global data on changes in soil carbon following afforestation, based on 43 previous studies. On average, they found a decrease in soil carbon in the upper soil layer (<30 cm) during the first five years after afforestation, with a recovery to previous soil carbon levels after 30 years.

5. *CONCLUSIONS*

The forest management practice options available to reduce emissions and/or increase carbon stocks can be grouped in four general strategies (adapted from Nabuurs *et al.,* 2007):

1. Maintaining or increasing forest area by reducing deforestation and degradation, and through increasing plantation areas or natural expansion of forest land (e.g., afforestation of abandoned lands).
2. Maintaining or increasing stand-level carbon density through application of appropriate silviculture techniques (e.g., thinning, partial harvests, species compositions, etc.).

3. Maintaining or increasing the landscape-level carbon density through forest conservation, longer rotations, fire management, and pest and disease control.
4. Increasing off-site carbon stocks in wood products and promoting forest-based products to substitute fuel and other materials (e.g., biomass, building materials, etc.)

In the future, climate change may impact forest growth responses dramatically and modify all the scenarios analysed here. In light of such uncertainty, adaptive management holds potential for developing adequate, operational forestry strategies in a world of constant social and ecological change (Nyberg, 1998). Increases in the frequency of both droughts and floods, or alterations in inter-annual rainfall distribution could have specific impacts. Although several climate forecasts indicate a generally positive effect on future forest growth (Sabaté *et al.*, 2002), local drought and changes in temporal and spatial rainfall distributions may make timber production and carbon storage difficult. Greater efficiency in use of resources has been reported in some studies, such as that of Bogino and Bravo (2014) for water in Mediterranean pines in Spain. The impact of climate change on forest growth,and the interaction of climate changes with silvicultural treatments (Olivar *et al.,* 2014), is differentiated in ecosystems across Europe (i.e., Bogino and Bravo, 2008, Bogino *et al.,* 2009, Olivar *et al.,* 2012, Granda *et al.*, 2014, Pretzsch *et al.,* 2014), The combined effects of reducing deforestation and forest degradation while promoting afforestation, forest management, agro-forestry and bio-energy have the potential to increase in the future (IPPC, 2007), contributing to climate change mitigation and sustainable development.

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