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Carbon pool dynamics after variable retention harvesting in *Nothofagus pumilio* forests of Tierra del Fuego

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Abstract

Background It is necessary to determine the implications for managing forest stands using variable retention harvesting for maintaining carbon and for calculating the effects of different harvesting practices on above- and below-ground carbon balance in forest ecosystems. In this context, forest carbon management has gained more attention among managers and policy-makers during recent years. The aim of this study was to determine carbon pool dynamics in different forest ecosystem components after variable retention harvesting (VRH) to characterize the ecological stability and quantify the recovery rate through the years-after-harvesting (YAH).

Methods Carbon pool compartmentalization of 14 different components was determined in 60 harvested and primary unmanaged forests during the first 18 YAH in Tierra del Fuego (Argentina). We compared them using uni- and multi-variate methods, relativizing the outputs with primary unmanaged forests.

Results We determined the effectiveness to retain carbon components in post-harvested stands under different retention strategies (aggregated vs. dispersed). The balance among carbon pool components changed between managed and unmanaged stands across the YAH, and was directly related to the impact magnitude. Aggregated retention improved the ecological stability of the harvested areas, where the below-ground components were more stable than the above-ground components. The recovery rate was directly related to the post-harvesting natural dynamics of the stands. The studied period was not enough to fully recover the C levels of primary unmanaged forests, but VRH showed advantages to increase the C pools in the managed stands.

Conclusions Promoting VRH can improve sustainable forestry at the landscape level and in the long term, generating positive synergies with biodiversity and the provision of ecosystem services. This study provides important new insights into forest carbon management, in particular to setting standards in carbon projects and sets the groundwork for analysing the economics of the mentioned harvesting systems.

Keywords Carbon reservoir, Temperate forest, Ecological stability, Recovery rate, Forest carbon management

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Introduction

Under international agreements, forests are increasingly recognized and promoted as major carbon (C) sinks of terrestrial ecosystems in order to mitigate climate change impacts (Pukkala 2018; Rogelj et al. 2019). However, forests are under natural and anthropogenic pressures (Marifatul Haq et al. 2022), which may create great variability in their effectiveness as carbon sinks. In this context, it is necessary to understand the main drivers that influence forest dynamics and how these factors are interacting in the context of global climate change and human infrastructure development (McDowell et al. 2020). Recent studies reported that direct (e.g. logging) and indirect (e.g. invasive species) human-derived actions generate biodiversity loss, land degradation, and major forest structure changes that greatly influence their ecological integrity, with only 40% of the natural forests still remaining functionally intact (Grantham et al. 2020). For example, after logging, natural forests shift to younger stands with greater homogeneity and less provision of ecosystem services (Perera et al. 2018). Understanding the drivers of vegetation dynamics is then critical for accurate prediction of global terrestrial ecosystem functions under future conditions, e.g. maintenance of forest carbon sink values.

In this context, new approaches of forest management are needed to mitigate these impacts through adaptation strategies, e.g. by conserving forest cover and maintaining the provision of ecosystem services at certain sustainable levels (Keith et al. 2014; Puhlick et al. 2016; Martínez Pastur et al. 2020). The different forestry strategies can alter the C stored pools and fluxes by modifying the stand net productivity (e.g. altering net C exchange with the atmosphere) (Fahey et al. 2010). One of these new approaches is named "forest carbon management" (FCM), and it is designed to improve carbon sequestration capacity and storage by adopting strategies that consider the long-term changes in forest C dynamics at the landscape scale (Ameray et al. 2021). Therefore, accurate carbon stock calculations have become of great importance for carbon accounting.

Forest carbon stocks are estimated using different proposals, and in general, with predetermined C conversion factors. The proposed methods include: (i) biome-average, where a single value of C per unit area is applied to entire biomes or forest type categories; (ii) forest inventory measurements to estimate C stock using allometric relationships, or (iii) remote-sensing tools where the uncertainty of C stock estimates can be high or low depending on the method chosen for modelling (Gibbs et al. 2007). C stock estimation tools can be valuable for reducing costs and capturing source variation at the landscape level (Jayathunga et al. 2018; Pötzschner et al. 2022; Silveira et al. 2022). However, accurate field C data and estimation is essential for modelling and validation. The use of conversion factors is widely employed in C inventories (e.g. 50% C in relation to tree biomass (Aalde et al. 2006), but this introduces large biases in the estimation of the forest C stock of the different components, with an overestimated error up to 9% in above-ground biomass depending on the forest type (Martin et al. 2018). For this, forest C accounting has led to the development of standardized guidelines for estimating more accurate C pools, promoting the capture of variation in included pools, thus improving monitoring accuracy, costs and efforts (Fahey et al. 2010).

The Intergovernmental Panel on Climate Change (IPCC) define five carbon pools, including above-ground biomass, below-ground biomass, dead litter, woody debris, and soil organic matter (Eggleston et al. 2006), but other pools such as understory vegetation can also be considered (Pearson et al. 2013). Above-ground biomass was the most dynamic C pool (Fahey et al. 2010) mostly affected by management practices and human-derived impacts (Angelstam et al. 2021). Usually, this pool was estimated by considering only stem tree volume (Jayathunga et al. 2018; Huang et al. 2020); however, other ecosystem components also contribute to more accurate C estimation, including branches, leaves, snag trees, shrubs, understory plants, and epiphytic plants (Peri et al. 2006, 2008, 2010; Tang et al. 2018; Sun and Liu 2020). Many of these components are reduced or eliminated under intensive management, despite their ecological importance in the natural cycles and ecosystem functions (Lindemayer et al. 2012; Guillaume et al. 2018), and must be considered in the forest C valuations. Below-ground biomass mainly includes live roots that influence C storage in different soil layers (Eggleston et al. 2006). Some studies included the estimation of large tree roots; however, the fine roots from trees and understory plants also contributed greatly to C storage (Peri et al. 2006; Peri and Lasagno 2010). This component estimation requires a lot of effort and time, but avoiding its inclusion in the C stock can lead to greater biases (Riutta et al. 2021). Although litter is not a significant pool component in forests (Vashum and Jayakumar 2012), it is closely involved in many ecosystem processes, including nutrient cycling, decomposition and detritivore traits (Uchida et al. 2005; Brousseau et al. 2019; Vivanco and Austin 2019). For this, their consideration in studies evaluating C stock contributes to accurate ecosystem function modelling. Coarse woody debris (CWD) is an important C pool component, but also an influential factor in many ecosystem functions, e.g. fauna habitat, runoff and erosion, and nutrient cycling (Campbell et al. 2019). Besides, most CWD estimates include only large pieces lying on the forest floor, and do not include the estimation of small pieces that were often integrated into the soil layers over long periods of time (Kimberley et al. 2019; Harmon et al. 2020). Finally, soil organic carbon (SOI) is the main component of C stock in forest ecosystems, but with great spatial variation (Scharlemann et al. 2014). The SOI estimation included the percentage of C, which was related to climate, topography, vegetation and soil origin (Bradley-Cook and Virginia 2018; Hounkpatin et al. 2021), but also soil density, which determines C content (Blake and Hartge 1986). In summary, accurate estimations of C stocks in forests require consideration of more components than those included in traditional C inventories and models.

Harvesting modifies forest structure and the other elements of the natural ecosystems, including the above- and below-ground components (Martínez Pastur et al. 2019). The most rapidly changing pool is generally above-ground live biomass, e.g. trees and understory plants, which are relatively easy to estimate (Fahey et al. 2010). However, harvesting also influences other C pools that are more difficult to measure directly, like root biomass, CWD, and SOI (Hume et al. 2018). An overall CO_2 net emission occurs after harvesting, mainly because of the changes in biomass and soils that should be considered in FCM and the development of new management and mitigation strategies (Scharlemann et al. 2014).

One harvesting alternative that combines economic and ecological dimensions is variable retention harvesting (VRH), which was designed to maintain in situ some of the provision of ecosystem services and biodiversity (Lindenmayer et al. 2012; Martínez Pastur et al. 2020). This management strategy has shown to improve biodiversity conservation, ecological cycles, habitat for avifauna, and regeneration dynamics, among others (Martínez Pastur et al. 2008a, 2009, 2013, 2019). Most of the research in VRH focused on forest cover persistence (e.g. remnant overstory stability and regeneration dynamics), as well as the impacts on biodiversity and other ecosystem services maintenance (see Soler et al. 2015, 2016). Because FCM gained more attention among managers and policy-makers during recent years, it is necessary to include more research in C components to develop more effective and precise decision-taking tools based on long-term research. In this context, the aim of this study was to determine carbon pool dynamics in different forest ecosystem components after VRH to characterize the ecological stability (resistance and resilience), and to quantify the recovery rate through the years-after-harvesting (YAH). We define the following specific objectives: (i) determine the carbon pool compartmentalization of the different components ($n = 14$) in harvested and primary unmanaged forests during the first 18 YAH. (ii) Determine the effectiveness of retaining carbon components in the post-harvested stands under different retention strategies (aggregated vs. dispersed) compared to primary unmanaged forests during the first 18 YAH. (iii) Determine the ecological stability of the different retention strategies (aggregated vs. dispersed) compared to primary unmanaged forests for the above- and below-ground carbon pool components, and quantify the recovery rate during the first 18 YAH. We define the following hypothesis: (i) the balance among carbon pool components changes between managed and unmanaged stands across the YAH, and is directly related to the impact magnitude. (ii) Aggregated retention improves the ecological stability of the harvested areas, where the below-ground components are more stable than the above-ground components due to logging activities. And (iii) the recovery rate is related to the post-harvesting natural dynamics of the stands and the retrieval of the main ecological process associated with the primary unmanaged forests.

Methods

Study area

Five forest landscapes dominated by pure stands of *Nothofagus pumilio* (Poepp. et Endl.) Krasser (commonly named lenga) were selected for samplings in Tierra del Fuego (Argentina). These forests were harvested with VRH across a YAH time-line: (i) Lenga Patagonia S.A. (54° 28′ 55″ S and 66° 49′ 21″ W, 2 YAH); (ii) Campo Chico ranch (54° 34′ 56″ S and 66° 54′ 80″ W, 4 YAH); (iii) Irigoyen Forest Reserve (54°36′32″ S and 66°36′15″ W, 9 YAH); (iv) Los Cerros ranch (54° 22′ 31″ S and 67° 51′ 48″ W, 12 YAH), and (v) San Justo ranch (54° 07′ 25″ S and 68° 35′ 13″ O, 18 YAH) (Fig. 1). VRH in Tierra del Fuego reserves 30% of the stand area in aggregates (AG, one circular patch per hectare of 30 m radius) and dispersed retention (DR, 10–15 m^2 ha^{-1}, representing 15–20% of the original basal area) homogeneously distributed between AG patches (Martínez Pastur et al. 2009, 2019). At each forest landscape, eight stands were selected (> 2 ha each): (i) four harvested, and (ii) four controls, defined as primary mature unmanaged forests without previous harvesting (PF). At each harvested stand the two retention types were sampled (AG and DR). The final design included 60 plots (3 treatments × 5 YAH × 4 replicates). The sampling design was not balanced in that YAH and study location co-occurred. To ameliorate this, PF and AR/DR values were compared considering the landscape influence (see statistical analyses), so the

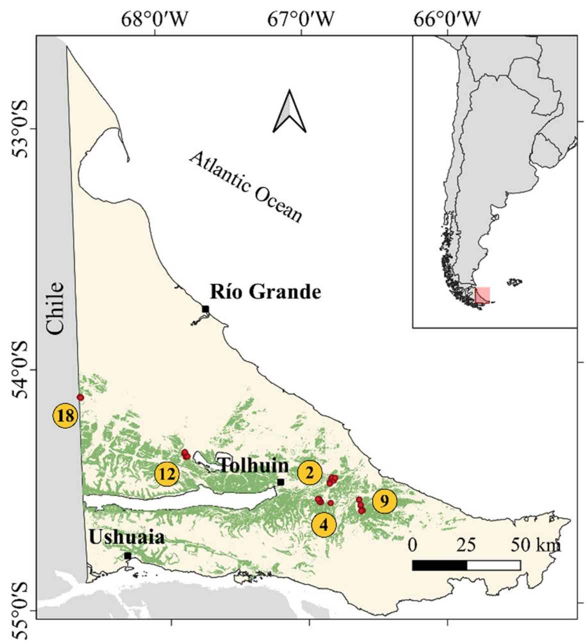

Fig. 1 Sampling forest landscapes and years after harvesting (2 to 18, orange circles), identifying the studied stands (red dots) in Tierra del Fuego, where *Nothofagus pumilio* forests are presented in green and cities in black squares

differences for any location could be considered when interpreting YAH impacts or relativized.

Data collection and measurements

At each plot, we randomly placed a 50-m transect during mid-summer (January to February). The transects were placed in the middle of AG patches (e.g. the centre of the transect and the centre of the aggregated were coincident), while transects occupied areas furthest away from aggregates in DR (e.g. parallel to the closest aggregate borders). Forest structure was characterized by two subplots located at the beginning and the end of each transect, using the point sampling method (BAF = 1–7) (Bitterlich 1984) with a Criterion RD-1000 (Laser Technology, USA). We measured all the live and dead trees, including the diameter at breast height (DBH) of trees > 5 cm with a forest calliper and the individual development phase for living trees (young trees for ages < 120 years, and mature trees for ages > 120 years). We also estimated the dominant height (DH) of the plots using a TruPulse 200 laser clinometer and distance rangefinder (Laser Technology, USA) by averaging the total height of the two tallest trees. These plots were complemented with the characterization of the advanced regeneration (> 1.3 m height and < 5 cm DBH) in 5 m^2 subplots. These data allowed us to determine: (i) the site quality (SQ) (1 the best to 5 the worst according to Martínez Pastur et al. 1997); (ii) the basal area of the overstory (BA, m^2 ha^{-1}) quantifying the proportion of mature (MBA, m^2 ha^{-1}) and young trees, as well as the advanced regeneration of tree saplings (SBA, m^2 ha^{-1}); (iii) total over bark volume of living (TOBV, m^3 ha^{-1}) and dead trees (DTOBV, m^3 ha^{-1}). We followed Martínez Pastur et al. (2002a) for modelling and calculations.

Understory cover was characterized using the point-intercept method (Levy and Madden 1933) with 50 intercept points (every 1 m) along each transect, measuring the diameter of each intercepted woody debris (> 1.0 cm diameter). To calculate the volume of coarse woody debris (CWD, m^3 ha^{-1}), we applied the line intersect sampling methodology (Marshall et al. 2003): (i) the volume was calculated using the measured diameter and an assumed cylindrical shape of 1 m length; (ii) the amount needed to cover a 1 m^2 surface was calculated, and (iii) we converted the CWD volumes in values per hectare. We also collected the above-ground biomass (tree seedlings and understory vascular plants) in a 0.25 m^2 subplot associated with each transect. Biomass was dried in an oven at 70 °C until a constant weight was reached, and the dry understory biomass of alive (LUP, t ha^{-1}) and dead (DUP, t ha^{-1}) plants was determined manually. Four soil samples (0–15 cm depth) were randomly collected along each transect using a 230.9-cm^3 field borer after previously removing the litter layer. Samples were weighed after air-drying under laboratory conditions (24 °C) to a constant weight, and soil was sieved with a 2-mm mesh, and when nodules or lumps of soil were found, they were broken down into their individual particles. Soil bulk density (SBD, t m^3) was obtained from the average of the four samples (soil weight over bored volume). For the analyses, the individual soil samples were pooled into one combined sample per transect. Organic matter (OM, %) was determined by weight loss after ignition in a muffle furnace at 500 °C for 24 h. The remnant elements (> 2 mm) were classified into fine roots (FIR, kg m^3 soil), wood and bark (WBS, kg m^3 soil), and rocks to calculate the corrected density as the weight of the sifted soil, divided for the borer volume minus the volume of roots, wood-bark and rocks contained in the samples.

Assumptions and carbon content modelling

We define four categories of forest carbon reservoirs: trees, deadwood, understory plants, and soil layer, including fourteen different components classified according to the ground (above- and below-ground (Table 1). We developed a user-friendly mathematical model based on species-specific primary (measured during field-work) data and applied it to a set of literature-based assumptions to adjust the precision of the calculation. For trees, we considered the following assumptions and

Table 1 Forest component of the defined reservoirs according to biomass allocation and categories

Category	Biomass allocation	Component	Abbreviation
Trees	Above-ground	Leaves	LEA
		Branches	BRA
		Bark	BAR
		Wood	WOO
	Below-ground	Coarse roots	COR
		Fine roots	FIR
Deadwood	Above-ground	Coarse woody debris	CWD
		Dead trees	DET*
	Below-ground	Coarse roots of dead trees	CRT*
		Wood and bark debris in soil	WBS
Understory plants	Above-ground	Live understory plants	LUP
		Dead understory plants	DUP
Soil layer	Below-ground	Litter	LIT
		Soil	SOI

*Dead trees (DT) = DET + CRT

models. TOBV of each alive tree was disaggregated into three different components (WOO = wood, BAR = bark, BRA = branches), while coarse roots (COR) were estimated as a proportion of TOBV, following Richter and Frangi (1992), Caldentey (1995) and Peri et al. (2006, 2008), and considering tree age (young or mature) and SQ of the stands. The volume of each component was transformed into biomass using an average basic wood density (0.551 kg m^3) (Atencia 2003; Barrientos Muñoz 2004; Schwarzkopf et al. 2018), and the carbon content of each component was calculated using estimates reported by Peri et al. (2010) and Chaves et al. (2022). Stand-level data were obtained for each modelled component (t C ha^{-1}) using the forest inventory plots measured during samplings. Leaves (LEA) were calculated based on litter production, as *N. pumilio* is a deciduous species. Annual leaf biomass was estimated for each treatment according to indexes obtained from long-term VRH plots (PF: 0.051, AG: 0.048, and DR: 0.093 t m^{-2} BA) (Martínez Pastur et al. 2008a, 2013), and where carbon content of leaves was estimated based on Chaves et al. (2022). Finally, fine root (FIR) biomass was determined using the weight of fine roots from sieved soil samples, borer volume, and the studied soil layer (0–15 and 15–30 cm). This biomass was multiplied by the carbon content reported by Peri et al. (2010). This FIR included trees and understory plants, which were not possible to split during sampling, and for further calculations both root types were combined in the tree category (Table 1). LEA and FIR data were finally presented at stand level (t C ha^{-1}).

For the deadwood category (Table 1), we considered the following assumptions and models. Carbon content of coarse roots (CRT) and stems of the dead trees (DET) were calculated using TOBV of dead trees from modelling proposed for living trees (basic wood density and carbon content). The volume of CWD was transformed into biomass by considering differential wood density values, which included a decay proportion due to natural decomposition. This decay in basic wood density was estimated using average values informed for other *Nothofagus* species (Stewart and Burrows 1994; Carmona et al. 2002; Coomes et al. 2002), due to the lack of information available for *N. pumilio*. A wood density decay of 55.6% was used for non-harvested areas (PF-AG) and 65.8% for harvested ones (DR). The volume of CWD was transformed into biomass using these calculated wood densities and multiplied by the carbon contents reported by Peri et al. (2010). Woody debris in soil (WBS) biomass was determined using the weight of woody debris from sieved soil samples, the borer volume, and considering the first 30 cm soil layer. This biomass was multiplied by the carbon content informed by Peri et al. (2010) and Chaves et al. (2022). With the forest inventory plots and modelled data, stand-level values were obtained for each modelled component (t C ha^{-1}).

For understory plants and soil layer values (Table 1), we considered the following assumptions and models. Understory biomass (LUP and DUP) obtained during the sampling were converted to carbon content (t C ha^{-1}) using the plant C concentration informed by the literature (Peri and Lasagno 2010; Ma et al. 2018). Litter (LIT) was calculated as the annual leaf (LEA) fall and the rate of decomposition in the two different environments (PF-AG and DR) according to Ibarra et al. (2011). The resulting values represent a litter biomass of × 1.22 annual leaf production in unharvested and × 0.55 in harvested environments. These values represent the regular litter layer that remains on the forest floor with different degrees of decomposition. The resulting litter biomass was multiplied by the carbon content reported by Peri et al. (2010) and Chaves et al. (2022). Finally, understory and litter values were calculated at the stand level (t C ha^{-1}). Organic matter (OM) determined in the soil samples was converted into soil organic carbon (SOI) content using a relationship obtained by regression ($n=25$ subsamples) between OM and the same samples quantified by an automatic analyser (LECO CR12, USA). Field values were transformed assuming a decay rate in carbon content along the soil profile, and an increasing soil bulk density (data not shown obtained from $n=120$ samples). Finally, SOI was multiplied by the SBD to obtain the soil carbon content (t ha^{-1}) for the first 30 cm depth layer.

Statistical analyses

Treatments were compared using uni- and multi-variate analyses considering different treatments (PF vs. VRH, AG vs. DR, PF vs. AG-DR, and the time-line defined by the five YAH values). VRH values were obtained as a combination of AG and DR multiplying the area occupied in the stand: 28.3% and 71.7%, respectively. Thus, 20 VRH stand values were obtained ($n=4$ VRH and 4 PF for each YAH period). We performed the following analyses: (i) one-way ANOVAs comparing the different control treatments (PF) obtained through the landscape (Additional file 1: Appendix 1). (ii) One-way (PF vs. VRH) and multiple (retention types × YAH) ANOVAs for the stand characteristics and biomass obtained during samplings (Table 2) and for the carbon contents obtained for different categories, biomass allocation and components (Tables 3, 4 and 5). (iii) One-way ANOVAs of above- and below-ground reservoirs, for different forests treatments (PF, AG, DR), and for VRH areas through the YAH periods (Fig. 3 and Additional file 1: Appendix 4). For these ANOVAs, we employed Fisher's test and Tukey's test at $p<0.05$ to separate means. Proportions of the different carbon reservoirs were plotted with R software using "ggplot2" package (Wickham 2016). (iv) We also performed principal component analysis (PCA) to compare and quantify the similarity among treatments according to the carbon content of the studied components (see Table 1), one comparing the forest treatments as a whole (PF, AG, DR) and discriminated by the YAH periods (Fig. 4 and Additional file 1: Appendix 5). PCA was set to calculate correlation coefficients among columns for the cross-products matrix, and assessed the axis significance with Monte Carlo permutation tests ($n=999$). Complementary, differences among plot groups were evaluated at each PCA by multi-response permutation procedures (MRPP) run with Bray–Curtis distances (Table 6). We used the statistics of MRPP to evaluate differences in general and between groups (McCune et al. 2002): t-statistic, which describes the split between groups (e.g. stronger separation for more negative t-values), and its associated p-value. Both multi-variate analyses (PCA and MRPP) were performed with PCORD 5.0 (McCune and Mefford 1999). Finally, (v) we measure the effect size using Hedges' g coefficient comparing primary unmanaged forests (PF) and stands under variable retention harvesting (VRH) classified according the different retention types (AG and DR) to determine the differences between treatments (Hedges 1981; Hedges and Olkin 1985).

Results

Comparison of the input variables among primary unmanaged forests and harvested stands

Differences among control unmanaged forests (PF) were tested before further comparisons were made to determine the potential influence of landscape effects. Primary unmanaged forests measured at the different locations did not present differences in most of the evaluated variables ($n=11$) used in the modelling (Additional file 1: Appendix 1), except for two variables directly related to soil characteristics (SBD and OM). SBD presented differences between control collected near harvested stand 4 YAH and 12–18 YAH, while OM presented differences between 4–9 YAH and 18 YAH. These lacks of differences for most of the components and the differences detected for the soil variables will be considered when we discuss the time-line results.

The comparison of the input variables between the primary forests (PF) and the harvested stands (VRH) showed differences in most of the above-ground variables, mainly those related to tree removal during harvesting, e.g. BA, MBA, TOBV, DTOBV, decreasing from PF to VRH stands. In contrast, SBA, CWD, LUP and DUP increased due to harvesting operations or due to changed environmental conditions (Table 2). Likewise, the same pattern was found between retention types inside the harvested stands (AG and DR), except for DTOBV, which did not present significant differences. Regarding the YAH, four variables had significant differences with two contrasting patterns. The first one showed an increase in the variable magnitude compared to PF, decreasing over time, and then recovering to the original values (e.g. DTOBV, OM, and DUP). The second pattern showed a decrease in the magnitude of the variable compared to PF, increasing over time until recovery of the original values (e.g. SBD). The variables associated with understory presented significant interactions, mainly associated with the different dispersion data between the plots growing in AG (low variability) compared to DR (high variability).

Table 2 Comparisons for the main stand characteristics used for modelling of the carbon content: (i) simple ANOVAs considering primary unmanaged forests (PF) and stands under variable retention harvesting (VRH), and (ii) multiple ANOVAs considering different retention types in harvested stands (AG: aggregated retention; DR: dispersed retention) and years after harvesting (YAH) as main factors

	Treatment	Level	DH	SQ	BA	MBA	SBA	TOBV	DTOBV	CWD	SBD	OM	FIR	WBS	LUP	DUP
(i)	Forest types	PF	21.34	3.27	54.71b	49.29b	< 0.01a	567.65b	56.15b	276.12a	0.53	20.92	6.78	2.77	0.51a	0.21a
		VRH	22.34	2.93	23.24a	21.54a	0.82b	258.62a	28.84a	496.89b	0.50	25.27	8.99	3.67	1.58b	1.14b
		F	2.24	2.40	141.2	86.71	10.66	63.26	8.17	14.03	0.14	1.80	3.14	3.16	42.51	15.39
		p	0.143	0.130	<0.001	< 0.001	0.002	< 0.001	0.007	< 0.001	0.715	0.188	0.084	0.083	< 0.001	< 0.001
(ii)	A: Retention types	AG	22.53	2.93	50.40b	45.60b	< 0.01a	556.20b	39.54	285.18a	0.47	25.07	7.65	3.12	0.71a	0.13a
		DR	22.26	2.94	12.53a	12.05a	1.15b	141.17a	24.62	580.45b	0.51	25.34	9.52	3.89	1.92b	1.54b
		F	0.17	< 0.01	137.53	67.39	12.87	125.19	1.67	16.24	0.72	0.01	1.50	1.46	32.70	35.80
		p	0.683	0.951	< 0.001	< 0.001	0.001	< 0.001	0.206	< 0.001	0.403	0.935	0.230	0.237	< 0.001	< 0.001
	B: YAH	2	22.71	2.84	34.63	34.63	< 0.01	398.89	21.51ab	434.13	0.70b	18.41a	11.39	4.65	1.19	0.82ab
		4	23.84	2.50	32.19	30.00	0.15	371.14	28.30ab	256.73	0.29a	30.61ab	11.19	4.57	1.59	0.54a
		9	21.39	3.22	28.06	16.81	1.08	288.24	66.26b	440.31	0.21a	41.99b	5.37	2.20	1.56	1.89b
		12	21.66	3.13	31.88	31.13	0.44	343.93	34.59ab	507.07	0.69b	15.90a	7.38	3.02	1.06	0.23a
		18	22.38	2.97	30.56	31.56	1.20	341.23	9.73a	525.82	0.56b	19.10a	7.60	3.10	1.16	0.70a
		F	1.78	1.55	0.44	2.30	2.30	0.98	2.71	1.68	18.40	9.20	2.34	2.30	1.07	5.76
		p	0.159	0.213	0.777	0.082	0.082	0.432	0.049	0.180	< 0.001	< 0.001	0.078	0.081	0.388	0.002
	Interactions AxB	*F*	0.64	0.47	0.34	0.45	2.31	0.12	2.11	1.54	0.88	0.77	1.34	1.36	3.29	4.60
		p	0.640	0.754	0.849	0.775	0.081	0.975	0.104	0.215	0.487	0.554	0.279	0.270	0.024	0.005

DH: dominant height (m), SQ: site quality, BA: basal area of living trees (m^2 ha^{-1}), MBA: mature trees basal area (m^2 ha^{-1}), SBA saplings basal area (m^2 ha^{-1}), TOBV: total over bark volume of living trees (m^3 ha^{-1}), DTOBV: dead tree total over bark volume (m^3 ha^{-1}), CWD: coarse woody debris (m^3 ha^{-1}), SBD: soil bulk density (t m^{-3}), OM: soil organic matter (%), FIR: fine roots (kg m^{-3} soil), WBS: wood and bark in soil (kg m^{-3} soil), LUP: live understory plants (dry weight t ha^{-1}), and DUP: dead understory plants (dry weight t ha^{-1})

F: Fisher test; *p*: probability. Different letters show significant differences in means using Tukey tests at $p < 0.05$

Table 3 Comparisons for carbon content (t C ha^{-1}) of total, biomass allocations and categories of forest reservoirs (see Table 1): (i) simple ANOVAs considering primary unmanaged forests (PF) and stands under variable retention harvesting (VRH), and (ii) multiple ANOVAs considering different retention types in harvested stands (AG: aggregated retention; DR: dispersed retention) and years after harvesting (YAH) as main factors

	Treatment	Level	Total	Above-ground	Below-ground	Trees	Deadwood	Understory plants	Soil layer
(i)	Forest types	PF	440.63a	245.38b	195.25	223.72b	63.64a	0.30a	152.98
		VRH	363.87b	177.60a	186.28	107.04a	101.28b	1.14b	154.41
		F	15.18	21.48	0.59	70.60	13.24	32.21	0.02
		p	< 0.001	< 0.001	0.466	< 0.001	< 0.001	< 0.001	0.893
(ii)	A: Retention types	AG	421.54b	236.67b	184.87	218.66b	60.35a	0.36a	142.18
		DR	341.11a	154.28a	186.83	62.99a	117.43b	1.45b	159.24
		F	15.97	31.40	0.02	135.79	17.41	44.5	2.41
		p	< 0.001	< 0.001	0.876	< 0.001	< 0.001	< 0.001	0.131
	B: YAH	2	433.21b	207.92	225.29b	161.14	88.03	0.84ab	183.20b
		4	346.59ab	170.90	175.70ab	151.49	59.80	0.91ab	134.41ab
		9	337.90a	186.17	151.73a	117.15	96.65	1.44b	122.67a
		12	401.20ab	208.42	192.78ab	137.41	102.47	0.55a	160.77ab
		18	387.73ab	203.97	183.76ab	136.92	97.52	0.78ab	152.51ab
		F	3.07	1.00	3.71	1.25	1.25	3.18	3.66
		p	0.031	0.421	0.014	0.312	0.312	0.027	0.015
	Interactions AxB	*F*	2.65	1.64	1.50	0.20	2.21	4.46	0.95
		p	0.053	0.189	0.226	0.939	0.092	0.006	0.452

F: Fisher test, *p*: probability. Different letters show significant differences in means using Tukey tests at $p < 0.05$

Table 4 Comparisons for carbon content (t C ha^{-1}) of the different components of trees category (see Table 1): (i) simple ANOVAs considering primary unmanaged forests (PF) and stands under variable retention harvesting (VRH), and (ii) multiple ANOVAs considering different retention types in harvested stands (AG: aggregated retention; DR: dispersed retention) and years after harvesting (YAH) as main factors

	Treatment	Level	LEA	BRA	BAR	WOO	COR	FIR
(i)	Forest types	PF	32.57b	19.39b	22.50b	113.80b	25.56b	9.89
		VRH	10.93a	8.88a	10.33a	52.13a	11.63a	13.13
		F	203.86	62.95	62.02	62.91	60.27	3.15
		p	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	0.084
(ii)	A: Retention types	AG	30.01b	19.00b	22.10b	111.52b	24.85b	11.2
		DR	3.40a	4.89a	5.68a	28.69a	6.41a	13.9
		F	201.62	124.35	116.50	123.48	145.23	1.50
		p	< 0.001	< 0.001	< 0.001	< 0.001	< 0.001	0.230
	B: YAH	2	17.66	13.62	16.05	80.07	17.07	16.68
		4	16.75	12.69	14.80	74.48	16.43	16.35
		9	15.58	9.92	11.00	57.92	14.95	7.78
		12	16.93	11.77	13.82	69.17	14.93	10.80
		18	16.61	11.73	13.79	68.91	14.79	11.10
		F	0.13	0.94	1.20	0.97	0.37	2.36
		p	0.972	0.455	0.332	0.439	0.827	0.075
	Interactions AxB	*F*	0.06	0.11	0.09	0.10	0.54	1.33
		p	0.993	0.979	0.985	0.981	0.709	0.283

LEA: leaves; BRA: branches; BAR: bark; WOO: wood; COR: coarse roots; and FIR: fine roots

F: Fisher test, *p* probability. Different letters show significant differences in means using Tukey tests at $p < 0.05$

Table 5 Comparisons for carbon content (t C ha^{-1}) of the components of the deadwood, understory and soil layer categories (see Table 1): (i) simple ANOVAs considering primary unmanaged forests (PF) and stands under variable retention harvesting (VRH), and (ii) multiple ANOVAs considering different retention types in harvested stands (AG: aggregated retention; DR: dispersed retention) and years after harvesting (YAH) as main factors

	Treatment	Level	CWD	DT	WBS	LUP	DUP	LIT	SOI
(i)	Forest types	PF	41.60a	17.62b	4.42	0.22a	0.08a	1.56b	151.41
		VRH	86.36b	9.05a	5.86	0.68b	0.46b	0.66a	153.75
		F	20.99	8.17	3.15	42.51	15.44	143.78	0.05
		p	< 0.001	0.007	0.084	< 0.001	< 0.001	< 0.001	0.827
(ii)	A: Retention types	AG	42.97a	12.40	4.99	0.31a	0.05a	1.53b	140.65
		DR	103.50b	7.73	6.21	0.83b	0.62b	0.32a	158.93
		F	24.65	1.67	1.50	32.5	35.9	150.89	2.76
		p	< 0.001	0.206	0.231	< 0.001	< 0.001	< 0.001	0.107
	B: YAH	2	73.85	6.75ab	7.43	0.51	0.33ab	1.01	182.19b
		4	43.62	8.88ab	7.30	0.69	0.22a	0.94	133.47ab
		9	72.35	20.79b	3.50	0.68	0.76b	0.84	121.83a
		12	86.82	10.85ab	4.81	0.46	0.09a	0.94	159.84ab
		18	89.52	3.05a	4.95	0.50	0.28a	0.90	151.61ab
		F	1.79	2.71	2.34	1.08	5.72	0.32	3.64
		p	0.158	0.049	0.078	0.385	0.002	0.864	0.016
	Interactions AxB	*F*	1.65	2.11	1.35	3.30	4.58	0.19	0.94
		p	0.188	0.104	0.276	0.024	0.005	0.941	0.454

CWD: coarse woody debris; DT: stems and coarse roots of dead trees (DET + CRT), WBS: wood and bark in soil; LUP: live understory plants; DUP: dead understory plants; LIT: litter; and SOI: soil

F Fisher test, *p*: probability. Different letters show significant differences in means using Tukey tests at $p < 0.05$

One study area (Campo Chico Ranch, 4 YAH) presented lower differences between AG and DR treatments compared to the other studied forest landscapes.

Changes in carbon content between primary unmanaged forests and variable retention harvesting stands

Total carbon stock was significantly higher in primary unmanaged forests (PF) than in variable retention harvested (VRH) stands, which presented a reduction at both biomass allocation categories (above- and below-ground) (Table 3). As expected, trees also decreased, and deadwood and understory plants increased due to harvesting operations, while the soil layer did not change significantly. Among the tree components (Table 4), all of them decreased in VRH (LEA, BRA, BAR, WOOD, COR), except for fine roots (FIR) which also included understory plants and did not present significant differences. Among the deadwood components (Table 5), CDW increased in harvested stands, while dead trees (DT = DET + CRT) decreased. Soil-integrated wood components did not differ significantly. Among the two understory plant components studied (LUP and DUP), both increased significantly due to harvesting. Finally, among the studied soil layer components, litter followed the same pattern as remnant overstory trees, which decreased in VRH, while soil carbon content did not change significantly between treatments.

The compartmentalization of the different carbon components changed with the harvesting (Fig. 2, Additional file 1: Appendices 2 and 3). The above-ground component was more important in primary unmanaged forests than in VRH (55.7% vs. 48.8%), where live trees and soil layer were the most important category in PF (42.7% + 34.7%), while soil layer and deadwood were the most important ones in VRH (42.4% + 25.9%).

Changes in carbon content between retention types and across YAH

Similar trends were found between AG and DR, where AG presented a similar response to PF. Total carbon stock was significantly higher in AG than DR, presenting a significant reduction in above- but not in below-ground components (Table 3). Trees, deadwood and understory plants (as a whole or discriminated by components) followed the same trends described before. These findings highlight the similarities between PF and AG, which can be also found when the different compartments were considered, e.g. AG and PF did not differ in the carbon contents of above- and below-ground biomass allocations. However, DR changed the pattern, where below-ground component became the most important

Table 6 Multi-response permutation procedure (MRPP) results comparing differences among groups of plots for carbon content (t C ha^{-1}) considering different forest treatments: (i) retention types as a whole, and (ii) retention types discriminated by years after harvesting from 2 to 18, where all primary unmanaged forests (PF) were used as control treatments for comparisons

		AG	DR	AG2	AG4	AG9	AG12	AG18	DR2	DR4	DR9	DR12	DR18
	PF	0.122 (0.449)	−23.214 (< 0.001)										
(i)	AG		−21.704 (0.311)										
	Overall	−10.731 (< 0.001)											
	PF			0.117 (0.468)	0.682 (0.730)	−0.555 (0.273)	0.163 (0.461)	0.617 (0.683)	−3.551 (0.005)	−3.518 (0.005)	−3.592 (0.007)	−3.630 (0.006)	−3.887 (0.005)
	AG2				0.886 (0.817)	−1.464 (0.085)	−0.217 (0.348)	1.044 (0.879)	−9.620 (< 0.001)	−10.394 (< 0.001)	−11.730 (< 0.001)	−11.122 (< 0.001)	−12.244 (< 0.001)
	AG4					−0.102 (0.380)	−0326 (0.349)	−1.016 (0.878)	−3.722 (0.006)	−4.145 (0.005)	−4.085 (0.006)	−3.991 (0.006)	−4.148 (0.005)
(ii)	AG9						−1.458 (0.077)	−0.615 (0.259)	−4.239 (0.006)	−4.288 (0.006)	−4.275 (0.006)	−4.169 (0.006)	−4.342 (0.006)
	AG12							0.454 (0.606)	−3.810 (0.006)	−3.981 (0.005)	−3.702 (0.008)	−3.807 (0.007)	−4.108 (0.006)
	AG18								−3.846 (0.006)	−3.805 (0.006)	−3.918 (0.007)	−3.791 (0.007)	−4.140 (0.006)
	DR2									−2.210 (0.036)	−2.650 (0.025)	0.368 (0.622)	−0.548 (0.263)
	DR4										−2.990 (0.013)	−1.895 (0.055)	−3.427 (0.008)
	DR9											−1.758 (0.063)	−2.268 (0.037)
	DR12												0.814 (0.784)
	Overall	−22.890 (< 0.001)											

AG: aggregated retention; DR: dispersed retention, *t*-statistic is shown with *p*-probability between brackets

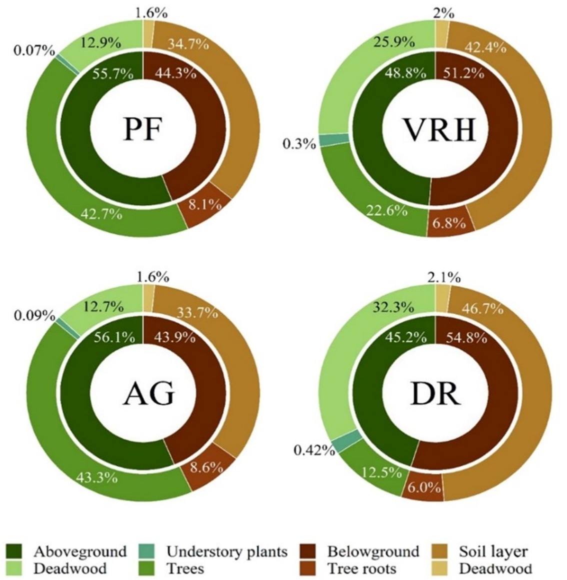

Fig. 2 Components of carbon content (%) classified according each category and biomass allocation (see Table 1) showing primary unmanaged forests (PF) and stands under variable retention harvesting (VRH) classified according the different retention types (AG: aggregated retention, DR: dispersed retention). The data are the mean of all the stands combining the years-after-harvesting samplings. Inner ring discriminates between biomass allocations (above- and below-ground), while outer ring shows category x biomass allocation sub-components

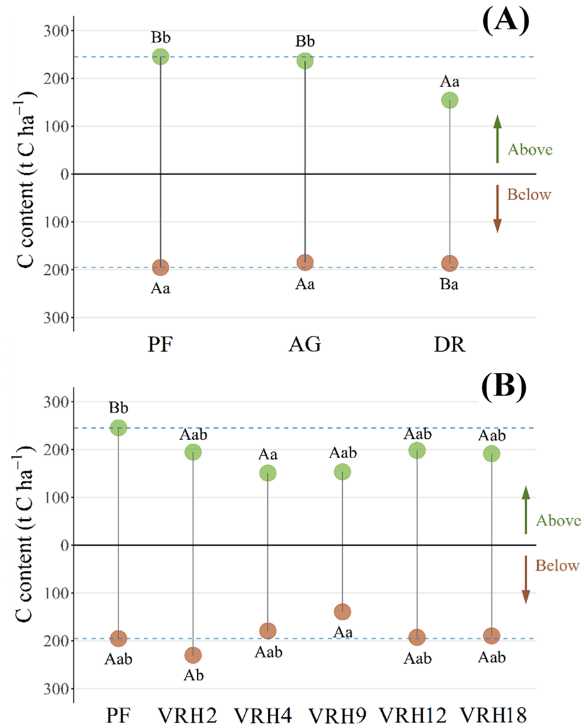

Fig. 3 Comparisons of carbon content (t C ha^{-1}) using simple ANOVAs: **A** primary unmanaged forests (PF) and different retention types in harvested stands (AG: aggregated retention, DR: dispersed retention), and **B** primary unmanaged forests (PF) and stands under variable retention harvesting (VRH) at different years-after-harvesting (2 to 18 years). Different letters showed significant differences between reservoirs above- (green) and below-ground (brown) (capital letters), or among treatments (lowercase letters). Dashed blue lines showed PF average carbon contents. Statistic values are presented in Additional file 1: Appendix 4

component for carbon stock (Fig. 3). The compartmentalization of the different carbon components changed between retention types (Fig. 2, Additional file 1: Appendices 2, 3 and 4). As well as PF, the above-ground component was higher in AG than in DR (56.1% vs. 54.8%), where live trees and soils were the most important components in AG (43.3% + 33.7%), while soils and deadwood were the most important components in DR (46.7% + 32.3%). These outputs influenced the PCA, where PF and AG plots were intermixed, and clearly split from DR plots (Fig. 4). Axis 1 highlighted the differences in tree components for PF + AG, and understory plants and CWD for DR (Additional file 1: Appendix 5). Axis 2 highlighted the differences in soil properties, which was identified as the main factor of change at the landscape level (Additional file 1: Appendix 1). MRPP analyses also identified the same trend, where PF and AG were similar, and significantly different from DR (Table 6).

When we compare the effect size using Hedges' *g* coefficient, we found negative (lower values in PF) and positive values (higher values in PF), which presented different effect sizes (Additional file 1: Appendix 6): (i) PF-VHR = most of the values of the above-ground presented large effect sizes (> 0.8), while below-ground components presented medium (0.5–0.8) to small (< 0.2) effect sizes; (ii) PF-AR = most of the values presented medium (0.5–0.8) to small (< 0.2) effect sizes, showing greater homogeneity between treatments; and (iii) PF-DR = most of the values presented medium (0.5–0.8) to large (> 0.8) effect sizes, showing greater heterogeneity between treatments.

Total and below-ground components carbon stock significantly changed across the YAH in the harvested stands (Table 3), decreasing between 2 and 9 years, and increasing from 9 to 18 years. However, when components were analysed year-by-year (Fig. 3), the same trend was detected for both biomass allocations, decreasing until year 9, and then increasing (e.g. soil layer also followed the same trend). Furthermore, trees and deadwood did not significantly change across the YAH (Table 3), but as expected, the understory layer significantly changed, following an opposite response. The understory plants increased until year 9, and then decreased until year 18. When we analysed specific carbon content components (Table 5), we found that dead trees (DT = DET + CRT) and DUP also increased until year 9, and then decreased,

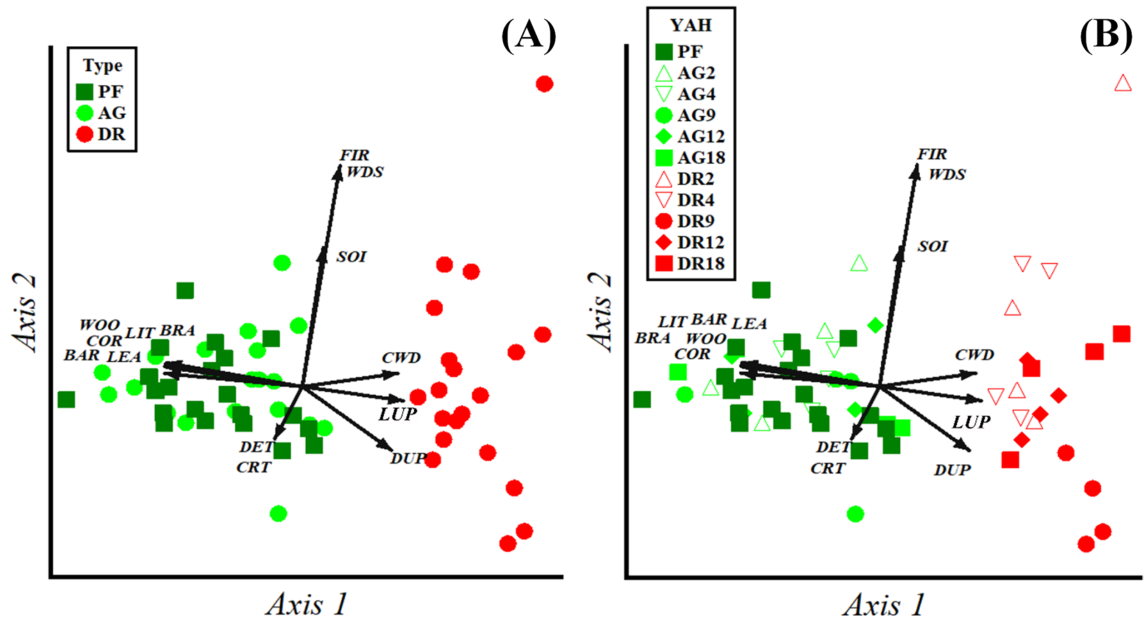

Fig. 4 Principal component analyses (PCA) for carbon content (t C ha^{-1}) considering different forest treatments: **A** primary forest (PF) and retention types (AG: aggregated retention, DR: dispersed retention), and **B** PF and retention types at different years-after-harvesting (2 to 18). Acronyms are shown in Table 1. Importance of each PCA component (Axis 1 and Axis 2) is detailed in Additional file 1: Appendix 5

while carbon content in soils followed an inverse pattern. These outputs can also be shown in the PCA. However, PF and the different retention types presented different behaviours (Fig. 4). AG plots were intermixed among the different biomass allocations and YAH showed that the carbon contents did not vary across the YAH for this retention type. The plot dispersion observed in AG was coincident compared to those observed in PF, without any trend in the plot distribution as a whole. However, in DR the plots followed the same pattern that was described before across the Axis 1. Closest DR plots belonged to YAH 2–4 and the most distant group belonged to YAH 9, and the YAH 12–18 group occupied an intermediate position. This pattern in Axis 1 was influenced by the tree components for PF + AG, and understory plants and CWD for DR (Additional file 1: Appendix 5). Once again, Axis 2 highlighted the differences in the soil properties influenced by the regional landscape. MRPP analyses also identified this trend and the observed differences (Table 6). PF and AG treatments (AG2 to AG18) did not present significant differences, but differed from all DR time-line treatments (DR2 to DR18). Aggregates did not differ among them, but all the AG treatments differed from DR treatments. These comparisons showed that AG carbon content stocks maintained similar values to PF; however, DR did not reach AG levels during the studied period (18 YAH). Finally, the comparisons among the different DR treatments showed three different groups: the first one integrated DR2 and DR18, which shared the DR12 plots with the second group that included the DR4. Last group, which presented significant differences with all the other treatments, was composed by the DR9. In brief, the studied time-line identified two stages: (i) one stage of post-harvesting which lost carbon content stocks (YAH 2 to 9), and (ii) a second stage which recovered the carbon stocks (YAH 9 to 18), but where the studied period was not long enough to reach the initial PF values.

Discussion

Modelling the forest components of the different C reservoirs

The proposed methodology considered the four most popular categories in research studies (e.g. Eggleston et al. 2006; Ameray et al. 2021), but included more components than usual ($n = 14$), both at above- ($n = 8$) and below-ground ($n = 6$). The trees category (representing 19–52% of the total C content in the different treatments) included more components ($n = 6$) than the others, followed by the deadwood category (14–34% of the total C content, $n = 4$ components), soil layer (34–47% of the total C content, $n = 2$ components), and understory plants (0.1–0.4% of the total C content, $n = 2$ components). For example, Coomes et al. (2002) and Gibbs et al. (2007) measured SOI, CWD, litter and some tree metrics.

The main advantage of this proposal was the greater detail in the tree component estimations (e.g. fine roots); however, these roots cannot be sorted from those belonging to the understory plants. For example, in primary

Nothofagus forests the understory cover is low, however, it acquires more importance (e.g. > 80%) in recently harvested stands (Martínez Pastur et al. 2002b; Argañaraz et al. 2020). Another advantage was the inclusion of the wood and bark debris integrated in the soil layer, which were rarely included in the C stock estimations. This component greatly influences C soil and other nutrient cycles (e.g. N) (Romero et al. 2005; Krzyszowska-Waitkus et al. 2006). This component represented between 5 and 8% of CWD in the studied stands, being one of the most important in C stock determination of forest ecosystems (Carmona et al. 2002; Campbell et al. 2019; Kimberley et al. 2019; Harmon et al. 2020). Finally, litter did not represent great amounts in C stock estimations (0.2–1.1% of the soil layer category), this variable has great ecological importance for nutrient cycling and soil biodiversity (Uchida et al. 2005; Vivanco and Austin 2019). Despite the advantages and weaknesses, improving the C stock estimations at stand level, this methodology allowed us to define accurate values in the databases for modelling at the landscape level, e.g. using remote-sensing tools (Pötzschner et al. 2022). To date, there are many new powerful methodologies that improve the estimation of forest cover, ecosystem functionality, degradation processes, and species composition (Köhl et al. 2015; Martin et al. 2018; Jayathunga et al. 2018; Grantham et al. 2020; Silveira et al. 2022). These variables can greatly influence C storage, and can be useful to be included in the development of new methodologies of C studies based on remote-sensing.

Sampling design and the influence of landscape in the input variables of primary unmanaged forests and harvested stands

The sampling design of this study covers most of the dispersion of the *N. pumilio* timber forests in Tierra del Fuego (Rosas et al. 2019). The sampling was conducted in one small area compared to the species distribution, and was made in the same site quality stands of pure forests (only one species in the canopy), with similar age composition (mature forests). For this, the differences can be mainly attributed to the management effect. However, the control treatment analysis is a key point to determine the influence of the landscape in the obtained results and modelling. In fact, one of the main strengths of the present study was the large number of sampled stands and the long-term data (e.g. 2 to 18 YAH), but the main weakness of the sampling design was the lack of replicates in the landscape (e.g. each YAH was sampled at one location) (Fig. 1). It was impossible to find all the time-line at every location due to sawmills are moving when the harvesting was finished. In Tierra del Fuego, the landscape can influence over soil types and depth layer, mainly due to the soil origin (areas covered or not by glaciers in the last ice age). The last glacial maximum (12.9 to 11.7 thousand years BP) varied from west to east, e.g. over soil development and stone percentage in the soil layer (Cheng et al. 2009; Palacios et al. 2020). However, these differences do not totally mask the main outputs and trends of the different treatments, e.g. the outputs follow some clear trends that are not related to the locations of the sampled stands. Other natural factors, that were not considered in the sampling design were: (i) age structure of the mature stands, where forests can be even-aged or uneven-aged (e.g. bimodal o irregular distribution) (e.g. see Martínez Pastur et al. 2021), and (ii) the influence of windstorms over the forest structure dynamics (e.g. promoting more presence of young trees in the overstory canopy due to local windthrow) (e.g. see Rebertus et al. 1997).

Besides the influence of the landscape, the differences in the input variables for the modelling between unmanaged (PF) and managed stands (VRH), and between retention types (AG and DR) were comparable to those variations previously reported, both during the short-term and the long-term periods after harvesting (Martínez Pastur et al. 2009, 2013, 2019; Lindenmayer et al. 2012). These trends can also show when we compare the effect size using Hedges' *g* coefficient, as was also cited by Soler et al. (2015, 2016) in meta-analyses studies of variable retention. The two described response patterns of the different values after harvesting (e.g. an increase or decrease magnitude of the values after harvesting) were also identified for managed *Nothofagus* forests in the long term (> 40 YAH) for forest structure variables and biodiversity values (Spagarino et al. 2001; Deferrari et al. 2001; Martínez Pastur et al. 2002b), but few studies considered the changes in environmental and abiotic variables in their analyses (see Martínez Pastur et al. 2019). Another key-study was related to the forest functionality that proposed the classifications of the stands in phenoclusters (Silveira et al. 2022), which resume the influence of many the landscape characteristics on the ecosystem functions. Our samplings were made in the same phenocluster main group, where the timber forests occurred.

Changes in carbon content between unmanaged and managed stands after harvesting

Harvesting reduces the stand C contents, both in the above- and below-ground components. Some components are more resilient than others, presenting different recovery rates. The tree category decreased in relation to the harvesting intensity, where AG maintained similar values to the control treatment (PF), showing the effectiveness in the short- and medium-term (18 YAH), as it was previously cited in papers that quantified the harvesting impact (Gea et al. 2004; Martínez Pastur et al. 2009; Soler et al. 2015). Deadwood increased in the managed stands by reducing the snag trees but increasing the CWD. This was mainly due to a common practice in Patagonia, where harvesting focused on high-quality timber logs, leaving low-quality logs in the managed stands after harvesting (Martínez Pastur et al. 2007). We expected changes in the wood components integrated in the soil, but they did not present significant differences. The understory category greatly increased due to the canopy opening, allowing plant regrowth and natural regeneration to recover losses in the overstory canopy over time (Martínez Pastur et al. 2011; Pérez Flores et al. 2019; Argañaraz et al. 2020). In contrast, the soil layer presented significant changes between unmanaged (PF + AG) and managed (DR) areas, but showed changes among YAH due to landscape effects as was discussed before. Other studies showed changes in the soil properties when different retention types were considered, where aggregates were more similar to controls than dispersed retention (Soler et al. 2015; Martínez Pastur et al. 2019). Jerabkova et al. (2011) and Kishchuk et al. (2014) did not find a clear relationship between VRH and C changes, however, they found other modifications in soil characteristics that can affect the soil biota and indirectly affect the C stocks (e.g. extractable NH_4-N, exchangeable K and Ca). These changes affected the compartmentalization of C stocks between control forests (PF) and managed stands. The above-ground C stocks were more important in PF (live trees and soils) than in VRH stands (soils and deadwood). This turnover in the C reservoirs has significant implications in the management strategies and provision of ecosystem services (Peri et al. 2006; Perera et al. 2018). In this context, the retention strategies allowed us to maintain some percentage of these values in the managed stands (e.g. AG and DR trees), improving the forest resilience (e.g. pests and climate extremes). The advantages of VHR were largely documented in the literature across different ecological functions (Lindenmayer et al. 2012; Martínez Pastur et al. 2020), as well as C reservoirs (Nunery and Keeton 2010; Zugic et al. 2021).

This study compared the first 18 YAH of the implementation of VRH in Tierra del Fuego (first cuts were conducted during 2000–2001 season), and did not cover the full rotation length of these temperate forest growing at high latitudes (e.g. forest term can be > 70–120 years) (e.g. Martínez Pastur et al. 2002a). For this, the studied period was not enough to reach the full recovery of the different attributes; however, the PCA showed the increasing of the similarities between the 18 YAH stands with the controls. The recovery in the forest structure values were comparable to those previously informed (e.g. Gea et al. 2004; Martínez Pastur et al. 2011; Soler et al. 2015, 2016), but our study also quantified these recoveries in terms of C content. In addition, our study allowed us to identify two stages: (i) one stage of post-harvesting which lost carbon stocks (YAH 2 to 9), and (ii) a second stage which recovered the carbon stocks (YAH 9 to 18), which probably continued the same trend across the years for a long term. For example, it was reported that periods > 200 years were necessary to recover the old-growth forest structure characteristics (Spagarino et al. 2001; Deferrari et al. 2001; Martínez Pastur et al. 2002b), where *N. pumilio* trees can live up to 450 years (Massaccesi et al. 2008; Matskovsky et al. 2019). Other studies also showed that the impacts of harvesting were extended from harvesting to the first ten YAH, and then the recovery of tree regeneration allowed to reduce the impacts in the following years (Gea et al. 2004; Martínez Pastur et al. 2011, 2019; Pérez Flores et al. 2019; Argañaraz et al. 2020).

Implications for the forest carbon management in Patagonia

Forest ecosystems store more than 80% of all terrestrial above-ground C and more than 70% of all SOI (Batjes 1996). This stock greatly declined during recent decades, and occurred mainly in living biomass and soils; however, forests also sequester about 30% of anthropogenic CO_2 emissions (Bellassen and Luyssaert 2014; Köhl et al. 2015; Ameray et al. 2021). In this context, maintaining and enhancing forest ecosystem carbon sequestration and storage is progressively becoming the main goal for sustainable forest management, and according to the Kyoto Protocol and United Nations Framework Convention on Climate Change is the best solution to face climate change (Ontl et al. 2019; Ameray et al. 2021). The silviculture proposed for *N. pumilio* forests is extensive forest management according to Ameray et al. (2021), which was defined as "the practice of forestry on a basis of low operating and investment costs per area", and involved partial cutting to promote natural regeneration (e.g. shelterwood cuts or VRH) (Gea et al. 2004; Martínez Pastur et al. 2019). For this, in contrast with intensive management (e.g. plantations), the extensive forest management based on moderate harvesting intensities and long rotations could be used not only to increase C storage, but also to provide other economic and ecosystem services (Perera et al. 2018; Tong et al. 2020; Ameray et al. 2021).

The forest carbon management could be based on two contrasting strategies (Ameray et al. 2021): (i) mature unmanaged forests should be left intact to conserve considerable quantities of C (Potapov et al. 2017); and (ii) young managed forests have higher C sequestration rate, where C storage can increase in the harvested wood products along their life cycle. Thus, total over bark volume and growth rate greatly varied in *N.*

pumilio forests (931 m^3 ha^{-1} and 1.1 m^3 ha^{-1} yr^{-1} in mature forests, and 115 m^3 ha^{-1} and 20 m^3 ha^{-1} yr^{-1} in younger managed stands) (Martínez Pastur et al. 2002b, 2008b), where timber products can reach 200 m^3 ha^{-1} every 70–120 years (Martínez Pastur et al. 2002a, 2009). The Intergovernmental Panel on Climate Change (IPCC) considers timber harvesting and products as a significant loss of C from forests, which remain outside the ecosystem boundaries for a variable period of time, e.g. C stored in furniture can remain sequestered for a very long time, while C in fuelwood results in an immediate emission (Gower 2003). Ameray et al. (2021) proposed three non-mutually exclusive strategies to mitigate global climate change, based on adaptive silviculture practices and extensive forest management (e.g. VRH) that would increase long-term ecosystem carbon sequestration and storage, in the natural ecosystems and the harvested wood products outside the managed stands: (i) preserving existing forest C stocks through conservation, mainly in the most valuable stands (e.g. old-growth forests); (ii) promoting extensive forest practices based on partial cuts, not only to increase natural forest productivity, but also preserving some of the existing forest C stocks in the managed stands (e.g. retention structures), and (iii) achieving high productivity in the managed areas through intermediate treatments (e.g. thinning) or more intensive practices (e.g. afforestation, fertilization, genomics). These forest management strategies must take into consideration all possible ecosystem services, from environmental, economic and social perspectives.

Variable retention harvesting (VRH) is gaining interest as one alternative to traditional management (e.g. clear-cuts or shelterwood cuts) to face climate change at the regional level, since they could enhance forest C sequestration and lower impacts on SOI (Zhou et al. 2013; Martínez Pastur et al. 2019; Zugic et al. 2021). The positive effect of these partial cuttings on forest C sequestration was directly related to the harvesting intensity. The different components greatly varied according to harvesting types (see Fig. 2), and were related to the YAH, e.g. understory and regeneration development significantly increased, while SOI decreased or did not vary across the years (Lee et al. 2002; Jandl et al. 2007; Pérez Flores et al. 2019; Argañaraz et al. 2020). The different VRH approaches can greatly influence C stocks in the managed stands (Lindenmayer et al. 2012), e.g. from aggregated retention and fires in Tasmania to root extraction of harvested trees in Sweden. SOI was less affected than above-ground biomass; however, the C loss was directly related to harvesting intensity (Jandl et al. 2007). For this, VRH can help to protect the SOI in the long term. These losses can be linked to C transfer from the litter (e.g. leaves, lifted branches, etc.), the stand conditions (e.g. micro-climate) (Martínez Pastur et al. 2013), as well as decomposition and soil respiration rates, which can increase following harvesting (Ameray et al. 2021).

Conclusions

Nothofagus pumilio can be managed with many harvesting approaches, e.g. from selective cutting to clear-cuts (Gea et al. 2004). However, the different approaches can greatly impact above-ground biomass and C stock. VRH combines different objectives (timber and conservation), but also it is a unique alternative for forest carbon management (FCM) strategies. The aggregates (AG) maintain the similar levels of C stock in the different studied components, reducing the impact over C levels observed in the dispersed retention (DR). VHR increases the resilience of the managed stands to potential climate variations and reduces the potential impact of micro-climate variations (e.g. aggregates influence over most of the cutting areas) (e.g. Martínez Pastur et al. 2019). In addition, nearly 40% of the forests are considered as protection areas by the Provincial Forest Law (145/94) in Tierra del Fuego, which store large C stocks in the unmanaged forests (Gea et al. 2004; Martínez Pastur et al. 2007). We can conclude that promoting VRH can improve the FCM at the landscape level and in the long term, which in combination with intensive management in the harvested areas (DR) with thinning can promote higher C sequestration in these forests. This proposal generates positive synergies with other biodiversity protections and provision of ecosystem services (e.g. water protection and nutrient cycles). Finally, it is necessary to improve the landscape effects that can mask some of the outputs where control treatments presented significant differences.

Abbreviations

ANOVA	Analysis of variance
AG	Aggregated retention
BA	Basal area
BAR	Bark
BRA	Branches
C	Carbon
COR	Coarse roots
CRT	Carbon content of coarse roots
CWD	Coarse woody debris
DBH	Diameter at breast height
DET	Stems of dead trees
DH	Dominant height
DR	Dispersed retention
DTOBV	Total over bark volume of dead trees
DUP	Dry understory biomass of dead plants
FCM	Forest carbon management
FIR	Fine roots
LEA	Leaves
LIT	Litter
LUP	Dry understory biomass of alive plants
MBA	Basal area of mature trees

OM	Organic matter
PCA	Principal component analysis
PF	Primary forests
SBA	Basal area of saplings
SBD	Soil bulk density
SOI	Soil organic carbon
SQ	Site quality
TOBV	Total over bark volume
VRH	Variable retention harvesting
WBS	Wood and bark in soils
WOO	Wood
YAH	Years-after-harvesting

Supplementary Information

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Additional file 1. Appendix 1. Analyses of the variance (ANOVA) for the main stand characteristics used for modelling of the carbon content comparing the primary unmanaged forests (PF) at the different studied locations (2 to 18 corresponded to the years-after-harvesting of the studied harvested areas, see Fig. 1). DH = dominant height (m), SQ = site quality, BA = basal area of living trees (m^2 ha^{-1}), MBA = mature trees basal area (m^2 ha^{-1}), TOBV = total over bark volume of living trees (m^3 ha^{-1}), DTOBV = dead tree total over bark volume (m^3 ha^{-1}), CWD = coarse woody debris (m^3 ha^{-1}), SBD = soil bulk density (t m^{-3}), OM = soil organic matter (%), FIR = fine roots (kg m^{-3} soil), WBS = wood and bark in soil (kg m^{-3} soil), LUP = live understory plants (dry weight t ha^{-1}), and DUP = dead understory plants (dry weight t ha^{-1}). **Appendix 2.** Pools carbon contents (%) according to their biomass allocation and category for forest types (PF = primary unmanaged forests, VRH = variable retention), retention types (AG = aggregated retention, DR = dispersed retention) and years-after-harvesting (YAH, from 2 to 18). **Appendix 3.** Components of carbon contents (%) classified according their category (see Table 1) for forest types (PF = primary unmanaged forests, VRH = variable retention), retention types (AG = aggregated retention, DR = dispersed retention) and years-after-harvesting (YAH, from 2 to 18). Acronyms are shown in Table 1. **Appendix 4.** Fisher test (*F*) and probability (*p*) of analyses of variance for carbon contents (t C ha^{-1}) comparing primary unmanaged forests (PF) and retention types (AG = aggregated retention, DR = dispersed retention), and variable retention harvested stands (VRH) through the years-after-harvesting (YAH, from 2 to 18), discriminated in above- and below-ground components (see Table 1). Graphical outputs are presented in Fig. 3. **Appendix 5.** Importance of each carbon component (see Table 1) for the Principal Component Analyses (PCA) considering different forest treatments (retention types and years-after-harvesting, YAH), analysing Axis 1 and Axis 2, where eigenvectors were scaled to unit length. **Appendix 6.** Measure of effect size using Hedges' *g* coefficient comparing primary unmanaged forests (PF) and stands under variable retention harvesting (VRH) classified according the different retention types (AG = aggregated retention, DR = dispersed retention). See codes and classification of components in Table 1.

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Availability of data and materials
The datasets used and/or analysed during the current study are available from the corresponding author on reasonable request.

Declarations

Ethics approval and consent to participate
Not applicable.

Consent for publication
The authors consent to publish the data included in this draft.

Competing interests
The authors declare that they have no competing interests.

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