Geoderma 403 (2021) 115265

Contents lists available at ScienceDirect

Geoderma

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

Changes in soil organic matter associated with afforestation affect erosion processes: The case of erodible volcanic soils from Patagonia

Ludmila La Manna [a,b,*], Manuela Tarabini [a,b], Federico Gomez [a,b,c], César Mario Rostagno [b,d]

[a] Centro de Estudios Ambientales Integrados, Facultad de Ingeniería, Universidad Nacional de la Patagonia San Juan Bosco (UNPSJB), Ruta 259 Km 16,4, 9200 Esquel, Chubut, Argentina

[b] Consejo Nacional de Investigaciones Científicas y Técnicas (CONICET), Argentina

[c] Instituto Nacional de Tecnología Agropecuaria (INTA), Esquel, Chubut, Argentina

[d] Instituto Patagónico para el Estudio de los Ecosistemas Continentales (IPEEC-CENPAT- CONICET), Bvard. Brown s/n 9120, Puerto Madryn, Chubut, Argentina

A R T I C L E I N F O

Handling Editor: Ingrid Kögel-Knabner

Keywords:
Pinus
Plantation
Rangelands
Soil erodibility
Land-use change
Soil carbon

A B S T R A C T

Large areas of the Patagonian Andean Region with high potential for planting fast-growing exotic conifers, based on the volcanic nature of its soils, are affected by erosion processes. This study aimed to analyse the effects of *Pinus ponderosa* afforestation on soil organic matter (OM) fractions in non-allophanic volcanic soils and to determine the relationship between organic matter, soil aggregates and erosion processes. The study was conducted along a forested hill slope showing different soil properties according to slope position. OM fractions and aggregate size fractions under different dispersion forces were analysed on 0–5 cm soil samples in rangelands and in 14- and 24-year old plantations on steep and gentle slopes. Simulated rainfall assays were performed to assess soil erodibility; OM and granulometry of sediments were also studied. Results showed that OM contents, mainly OM labile fractions (i.e., particulate OM, and OM associated with macro and large microaggregates), soil aggregation and the formation of very stable microaggregates were enhanced in the afforested soils. However, soil changes varied depending on the initial soil OM contents and on the age of the plantation, with more erratic and smaller changes in most fertile soils. Although potential erosion rates are lower in plantations than in rangeland soils, the high OM enrichment rates found in sediments imply a high OM loss when the soils remain uncovered. Erosion processes in afforestation involve the removal of microaggregates rich in OM and silt fractions, while in the rangelands, coarse and very coarse sand single particles are lost. Afforestation replacing degraded rangelands may be a way to control erosion in these highly erodible volcanic soils, as long as the soil remains covered. Otherwise, the loss of soil enriched in OM from the superficial soil could favour soil carbon depletion.

1. Introduction

Throughout the Patagonian Andean Region of Argentina, soils are mainly developed from volcanic ash. The subhumid sector, a transitional area (ecotone) between the forest and the steppe, has suffered the highest human pressure in this region: traditionally, from extensive sheep- and cow-grazing, and more recently, from feed-lots, agriculture and expanding cities. Recent studies conducted in the ecotone of Chubut province, based on fallout radionuclides (Caesium-137), showed that soil losses in the last 50 years under different land uses were as high as 33 Mg ha^{-1} $year^{-1}$ (La Manna et al., 2019). Water erosion studies proved that volcanic soils are highly erodible when they lose their vegetation or litter cover (Rodríguez Rodríguez et al., 2002; La Manna et al., 2016). In the ecotone of the Andean Region, coarse-textured soils (loamy sand to sandy loam) are highly erodible (La Manna et al., 2016). In volcanic soils the sand fraction has a high content of volcanic glass, which is strongly vesicular or pumiceous, and can be considered a light fraction (McDaniel et al., 2012; Nanzyo and Kanno, 2018). Besides, the erosive processes in volcanic soils involve the removal and transport of large aggregates, due to the impact of raindrops (Rodríguez Rodríguez et al., 2002).

Given the high forest potential of volcanic soils, afforestation with fast-growing conifers is promoted as a profitable activity, an alternative to extensive livestock production, as well as a measure to mitigate erosion (Irisarri and Mendía, 1997). Simulated rainfall assays showed negligible values of erosion in forest soils completely covered by litter (La Manna et al., 2016). Pine plantations can also reduce soil erosion due

* Corresponding author at: Centro de Estudios Ambientales Integrados, Facultad de Ingeniería, Universidad Nacional de la Patagonia San Juan Bosco (UNPSJB), Ruta 259 Km 16,4, 9200 Esquel, Chubut, Argentina.

E-mail address: llamanna@unpata.edu.ar (L. La Manna).

https://doi.org/10.1016/j.geoderma.2021.115265
Received 11 August 2020; Received in revised form 6 April 2021; Accepted 20 April 2021
Available online 17 June 2021
0016-7061/

to the rainfall interception by the canopies, varying according to stand age, species and forest management (Buduba, 2006; Zhou et al., 2002; Gomez et al., 2019). However, many studies around the world question the ability of pine plantations to control erosion (e.g., Gu et al., 2013; Cao et al., 2015) and highlight their influence on the increase of fire intensity and frequency (Raffaele et al., 2016). Plantations are also controversial due to their possible negative effects on soil quality (Amiotti et al., 2000; Berthrong et al., 2009).

Land-use change towards afforestation with exotic conifers notably favours the deposition of great amounts of slowly decomposing leaf litter (Buduba et al., 2017), attributable to the cold climate and the high lignin/nitrogen ratio (Mazzarino et al., 1998). In SE Spain, Peinado et al. (2016) found that soil organic carbon significantly increased in pine plantations in comparison with the unplanted, degraded areas (0–5 cm). However, not always does afforestation improve soil quality in the short term. According to Davis and Condron (2002), the substitution of grasslands with planted forests usually leads to an initial decline in soil organic matter, probably related to distinct strategies in carbon allocation and consequent changes in cycling (Six et al., 2002a).

Soil carbon generally decreases in afforestation in the first five years, followed by a decrease in the rate of decline and later recovery, with plantations older than 30 years reaching values similar to those found under the previous agricultural systems (Paul et al., 2002). In Patagonia, there is no evidence to support an increase in carbon sequestration in plantation soil in comparison with grassland (Laclau, 2003; Laclau et al., 2008; Buduba et al., 2017). Furthermore, organic matter significantly decreases where conifer plantations replace native forests (Candan and Broquen, 2009). However, pine afforestation replacing degraded rangelands on non-allophanic volcanic soils, with low values of organic matter, showed significant increases in organic matter contents (La Manna et al., 2018). These results suggest that afforestation age is not the only important factor for understanding changes in organic matter because some initial soil properties, which could control the change direction, are equally important. In volcanic soils, the presence of non-crystalline minerals (allophane/imogolite) and organic matter contents are key variables for determining soil fertility (McDaniel et al., 2012). Both variables control water storage capacity (Dahlgren et al., 2004), which is crucial in an area where rainfall is out of phase with the growing season.

The stabilization of organic matter in soil aggregates in Andisols can vary depending on land use (Lu et al., 1998), and it is unknown how the plantations affect the organic matter fractions in volcanic soils of Patagonia. For example, studies in other parts of the world have shown changes in particulate organic matter associated with plantations, but not in total organic matter (Mao and Zeng, 2010). In contrast, other studies found no clear evidence that the more labile organic matter fractions were more sensitive indicators of the impact of plantations than total organic matter (Lima et al., 2006). The accumulation of organic matter in forest soils depends on many factors, including climatic conditions, soil properties, soil moisture, plant cover and forest management (Błońska and Lasota, 2017; Plaza et al., 2018).

Different studies suggest that forest soils have better aggregation than soils under other vegetation (Rienzi et al., 2018). Conversion of cultivated land to pine plantations resulted in increased aggregation and greater intra-aggregate particulate organic matter, suggesting that this organic matter fraction is the most relevant in forested soils (Six et al., 2002a). On the other hand, studies on agricultural soils suggest that the free light fraction of organic matter is stabilised to a greater extent in volcanic than in non-volcanic soils, and that the intra-aggregate fraction plays a minor role (Zagal et al., 2013). Thus, the effect of land-use change on organic matter fractions seems to greatly vary according to the type of soil and the fractionation technique (Poeplau et al., 2018).

The objectives of this study were to analyse the effects of plantations on soil organic matter fractions in coarse-textured non-allophanic volcanic soils and to determine the relationship between organic matter, soil aggregates and erosion processes.

1.1. Hypotheses

The effect of pine plantations on the upper centimetres of the soil is positive, and the magnitude of this effect varies depending on plantation age and on the initial soil organic matter contents; positive changes are expected to be greater in less fertile soils. The erosion processes in forested soils involve large losses of organic matter.

2. Materials and methods

2.1. Study area

The study was conducted along a forested hill slope in the Percy River Basin, Chubut province, in the forest-steppe ecotone of the Patagonian Andean Region (Fig. 1). Mean annual precipitation is ca. 700 mm, concentrated mainly in autumn and winter. Soils, classified as Entic Haploxerolls, are developed from volcanic ash and they show slight development of horizons, with sequences A/AC/C, and variable organic layer cover and thickness according to vegetation (Table 1). Soil profiles, laterally homogeneously distributed, have loamy sand textures and halloysite as dominant clay, resulting in highly erodible soils (La Manna et al., 2016).

Pinus ponderosa Douglas ex Lawson was planted all along the hill slope, replacing degraded rangelands. Pines were planted 14 and 24 years ago. This study focused on two slope positions: the maximum slope in the upper part (moderately steep slope, ca. 25%) and a gentle slope (ca. 9%), towards the bottom (Fig. 1). According to the slope, soil cover differed between rangelands and plantations (Table 1). On the steepest sector, soil cover in the rangeland was low, and the soil was mostly bare or covered with medium-sized gravels. On the gentle slope, although bare soil remains, litter and herb cover predominated, showing a better ecological condition. The dominant grasses were Festuca sp. (v.n. coirón). The dominant shrubs and half shrubs were *Molinum spinosum* (Cav.) Pers. (vn.neneo) and *Acaena splendens* Hook. &Arn, both indicative of degradation and overgrazing (Beeskow et al., 1995). Soil cover also differed between plantations according to the slope position. While a low percentage of bare soil remained on the steep-slope plantations, soil was completely protected on the gentle slope, by either litter or vegetation cover. In each sector, litter cover and organic horizon thickness increased, and understorey cover decreased with plantation age (Table 1).

2.2. Soil sampling and analysis

In each study site (i.e., rangeland, young afforestation and mature afforestation on the steep and on the gentle slope), systematic sampling was carried out and six sampling points, placed ca. 10 m apart from each other, were selected. The first sampling point was chosen at random. At each sampling point, soil cover and the organic horizon thickness were visually determined.

Soil hydrophobicity and aggregate stability were assessed in situ on the mineral soil. Soil hydrophobicity was assessed using an ethanol–-water solution, with different ethanol concentrations, and classified according to Doerr (1998). Aggregate stability was determined following Herrick et al. (2001), on surface (0–5 cm) and sub-surface (5–10 cm) soil samples. Since sampling was done in late summer, with dry conditions, field tests were performed under comparable soil moisture conditions (around 5%).

At each sampling point, a 0–5 cm mineral undisturbed soil core sample was taken with an auger, for bulk density and porosity estimation. A 0–5 cm mineral soil bulk sample was also taken, air-dried and passed through a 2 mm sieve. Electrical conductivity (EC) (soil–water ratio 1:2.5), pH in water (1:2.5) and pH in NaF 1 N (1:50) were measured. pH NaF measured at 2 and 60 min is considered an indicator of non-crystalline aluminosilicates in soils (Fieldes and Perrot, 1966).

Total organic matter (OM) was analysed by loss on ignition (IRAM-

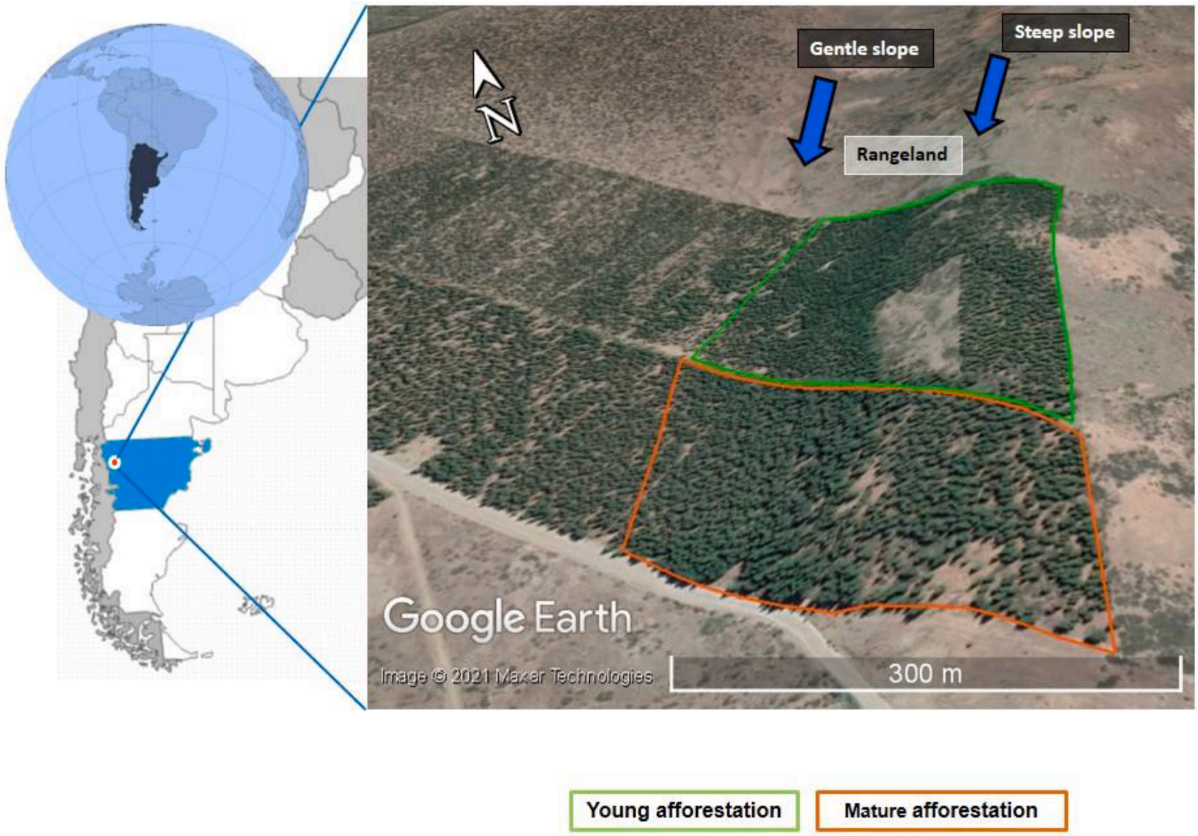

Fig. 1. Location of the study area in Chubut province, Argentina, and detail of the study area: rangeland, young and mature plantations on gentle and steep slopes.

SAGPyA, 2008). Physical fractionations of the soil OM were performed by the particle size technique. A procedure with a previous chemical dispersion was used to determine particulate OM (POM), and another one without chemical dispersion was used to determine OM associated with water-stable aggregates of different sizes.

POM, defined as OM associated with the fraction retained on a 53 μm mesh sieve after dispersing the soil (Cambardella et al., 2001), was determined following Videla et al. (2008), adapted from Cambardella et al. (2001). The 2 mm-sieved samples were dispersed with sodium hexametaphosphate (5 g L-1), stirred for 18 h on a horizontal shaker at 150 rpm, and finally, wet sieved on a 53 μm mesh sieve. The fraction retained on the sieve (>53 μm) was oven-dried at 100 °C for 24 h and weighed, and POM was analysed by loss on ignition (IRAM-SAGPyA, 2008).

OM associated with water-stable aggregates (OM_{Agg}) was also determined on 2 mm-sieved samples, for comparison with POM. Soil samples were placed on the top of a 250 μm sieve and submerged for 5 min in deionised water. Sieving was manually done by gently moving the sieve in four directions (forward, backward, left, right) for 2 min, to achieve aggregate separation, based on Six et al. (1998). The fraction retained on the sieve (F > 250) was oven- dried and weighed, and $OM_{>250}$ was analysed by loss on ignition (IRAM-SAGPyA, 2008). The soil particles that passed through the sieve (i.e., <250 μm) were processed in the same way on a 53 μm sieve, and the proportion of 53–250 μm fraction (F53-250) and its OM content ($OM_{53\text{-}250}$) were determined. OM associated with water-stable aggregates (OM_{Agg}) was calculated as the average between $OM_{>250}$ and $OM_{53\text{-}250}$ weighted by the weight of fractions.

For both analytical procedures (POM and OM_{Agg}), a soil fraction with particles smaller than 53 μm was washed and discarded, and $OM_{<53}$/OM were considered to be equal to 1-(POM/OM) and to 1- (OM_{Agg}/OM) (Enriquez and Cremona, 2018), corresponding to mineral associated silt- and clay-organic matter (Cambardella and Elliott, 1993) for procedures with and without previous chemical dispersion, respectively.

The proportion of the material retained on the sieves, with particles between 53 μm and 2 mm, was defined as Fd > 53 (for the samples previously dispersed by sodium hexametaphosphate) and F > 53 (for the wet-sieved samples without dispersion). The proportions were determined with respect to the total soil sample used for fractionation. The dispersed soil fraction with particles smaller than 53 μm was washed and discarded, and it was considered that Fd < 53 = 1 - Fd > 53 and F < 53 = 1 - F > 53, following Enriquez and Cremona (2018).

Grain-size distribution was analysed using a Malvern laser granulometer after destruction of OM with H_2O_2 at 80 °C, dispersion with sodium hexametaphosphate 1%, and ultrasound. Since Malvern equipment (Model 3000E) analyses particles smaller than 1000 μm, coarse sand fraction was separated by sieving. The fraction < 53 μm was defined after complete dispersion (i.e., OM-free, chemically dispersed and ultrasounded) as Fcd < 53, calculated as the proportion of clay + silt fractions and corrected by OM content, since it was removed.

2.3. Rainfall simulation experiments

Rainfall simulation experiments were performed adjacent to four of the six soil sampling points described above. In the rangelands, plots for the rain experiment were placed in sites representative of the average mean cover (see Table 1). In the plantations, rainfall assays were conducted in two conditions: i) unmodified soil and ii) removal of fresh litter and organic horizon, with mineral soil exposed. The organic layer (i.e., fermentation layer) refers to a layer of slightly to highly decomposed needles, fine twigs, and other organic material. The removal of non to highly decomposed plant remains resulted in an artificial soil surface condition that allowed us to analyse the bare mineral soil erodibility. Litter, vegetation and rock fragment cover were visually estimated by vertical projection in each sample plot before rainfall application.

Table 1
Topography and cover of surface (%) of the study sites.

Sector	Slope (%)	Rangeland					Young afforestation							Mature afforestation						
		Bare soil	Gravel	Litter	Shrub and half shrub	Herbs	Bare soil	Gravel	Litter	Shrub and half shrub	Grass	Fresh litter layer thinckness (cm)	Organic horizon thickness (cm)	Bare soil	Gravel	Litter	Shrub and half shrub	Grass	Fresh litter layer thinckness (cm)	Organic horizon thickness (cm)
Steep-slope	25	64	31	1	3	1	7	0	62	21	10	2.1 ± 0.1	2.0 ± 0.3	7	4	76	10	3	4.7 ± 1.2	3.1 ± 1.1
Gentle-slope	9	31	0	30	4	35	0	0	83	1	16	1.9 ± 0.6	0.5 ± 0.3	0	0	100	0	0	4.0 ± 0.5	3.3 ± 0.4

Table 2
Soil properties of rangelands on different slope positions.

	OM[a]	OMAgg[a]	POM[a]	$OM_{>250}$/OM[b]	$OM_{53\text{-}250}$/OM[b]	POM/OM[b]	$F^c_{>250}$	$F^c_{53\text{-}250}$	$Fd^c_{>53}$	VCS [d]	CS [d]	MS [d]
Steep-slope	3.32±0.22*	2.44±0.36*	1.33±0.20*	0.16±0.04*	0.32±0.03	0.38±0.06*	0.29±0.02*	0.47±0.02*	0.66±0.04*	6,8±0,65*	6,51±1,27*	13,52±1,19
Gentle-slope	7.48±1.13*	6.79±1.42*	6.88±0.56*	0.39±0.05*	0.45±0.03	0.79±0.02*	0.51±0.03*	0.33±0.02*	0.82±0.01*	14,86±1,21*	11,08±4,05*	11,97±4,45
	FS [d]	VFS [d]	Silt	Clay	pH	EC [e]	pH NaF 2‘	pH NaF 60‘	Porosity (%)	Hydrophobicity [f]	Aggregate stability surface [g]	Aggregate stability subsurface [g]
Steep-slope	22,65±0,53	16,14±0,86	36,2±1,07	4,77±0,60*	6.21±0.06	59.10±8.41	7.94±0.05	8.71±0.05	53.53±1.41*	8.5 (3–13)*	2 (2–2)*	5 (5–5)*
Gentle-slope	18,2±1,09	17,13±2,26	39,3±6,49	1,8±0,38*	6.00±0.07	79.23±11.04	8.53±0.23	9.05±0.16	63.36±2.02*	13 (8.5–13)*	0 (0–0)*	6 (6–6)*

Asterisks indicate soil variables that significantly differ between rangelands on different slope position.
a Organic matter contents (%): OM: total organic matter; OMAgg: organic matter associated with stable aggregates; POM: particulate organic matter.
[b] Organic matter fractions (proportion): $OM_{>250}$/OM: OM associated with stable macroaggregates with respect to total OM; $OM_{53\text{-}250}$/OM: OM associated with stable large microaggregates with respect to total OM; POM/OM: POM with respect to total OM.
[c]Aggregates size fractions: F > 250: proportion of water stable macroaggregates; F53-250: proportion of water stable large microaggregates; $Fd_{>53}$: proportion of stable aggregates after chemical dispersion treatment.
[d]Granulometric fractions (%): VCS: very coarse sand; CS: coarse sand; Ms: medium sand; FS: fine sand; VFS: very fine sand. The size fraction were determined by laser diffraction, expect VCS, which was separated by sieving.
[e] Electrical conductivity (μSm^{-1}).
[f]Hydrophobicity according to "Molarity of an Ethanol Droplet" technique: 0%=Very hydrophilic; 3%= Hydrophilic; 5%= Slightly hydrophobic; 8.5%= Moderately hydrophobic; 13%= Strongly hydrophobic; 24%= Very strongly hydrophobic; 36%= Extremely hydrophobic (Doerr 1998).
[g]Aggregate stability classes range from 0 = Soil too instable to measure; to 6 = Strong stability (Herrick et al. 2001).

A drip-type rainfall simulator was used to simulate rainfall on 1.564 cm^2 plots, following the methodology described by La Manna et al. (2016). Water with low solute concentration (E.C. $\leq$ 148 μS cm^{-1}) was used for the assays. The applied rainfall intensity was 100 mm h^{-1} for 30 min. Drop diameter of the simulated rainfall was 2.5 mm. Falling from 2.1 m, these drops reached a velocity of 5.3 ms^{-1}. Runoff was collected and measured at 5-min intervals in 1-L containers. Sediment production was determined from the total runoff collected for each simulation. The sediment production was obtained after suspended solid decantation (72 h) by carefully discarding most of the runoff. Sediments were collected in 500-mL beakers, dried for 48 h at 60 °C and weighed.

Sediments were analysed for OM content by the loss-on-ignition method (IRAM-SAGPyA, 2008). OM loss was calculated as the product of sediment yield and sediment OM content. The enrichment ratio (ER) of OM was calculated by dividing the content of OM in the sediments by its content in the original soil material (Avnimelech and McHenry, 1984). This variable allowed us to determine whether the sediments were enriched with OM, as compared to the contributing soils. Sediment texture was analysed by Malvern laser granulometer after destruction of organic matter with H_2O_2 at 80 °C, dispersion with sodium hexametaphosphate 1%, and application of ultrasound, while the coarse sand fraction was separated by sieving.

2.4. Data analysis

To characterise the variability of soils according to slope and vegetation type, a discriminant analysis was performed, considering each study site as a group (i.e., rangeland, young afforestation and mature afforestation on the steep and on the gentle slope, as dependent variables). All the measured soil properties were included at first as independent variables, and a stepwise procedure was used to retain the most significant variables for discrimination. Stepwise discriminant analysis is an efficient procedure for removing redundant variables and selecting those of greatest discrimination capacity (Afifi and Clark, 1984). The classification rates were assessed by a cross-validation procedure.

Data were analysed by parametric or non-parametric tests depending upon whether data satisfied normality and homogeneity of variances assumptions. Parametric test assumptions were checked by Q-Q plots, Shapiro-Wilk's W test (modified) and Levene's test, following Di Rienzo et al. (2013). Differences in soil properties between rangelands on steep and gentle slope were analysed by Kruskal–Wallis test. To analyse soil differences between the study sites, two-way analyses of variance (ANOVA) were carried out considering sector (gentle/steep slope) and vegetation type (rangeland/young afforestation/mature afforestation) as factors. Since both factors showed significant interaction for most variables, one-factor ANOVA was performed for each sector, considering the vegetation type as factor. Duncan test was applied as post hoc analysis. Categorical variables (i.e., hydrophobicity and aggregate stability) were analysed by Kruskal–Wallis test and pairwise comparisons. Wilcoxon, a non-parametric test for paired samples, was performed for analysing differences between Fcd < 53, Fd < 53 and F < 53, within each study site.

A similar procedure was applied to the analysis of sediment yield and runoff data from the simulated rainfall assays (i.e., ANOVAs and Duncan test, considering sector and rain treatments as factors). Differences in OM losses and sediment enrichment ratio between rain treatments were analysed by Kruskal–Wallis test and pairwise comparisons for each sector. The different particle size fractions between the soil and the sediments were compared by paired sample *t*-test. Correlations between variables were performed by Pearson test.

The analyses were carried out with the Infostat software (Di Rienzo et al., 2013) and SPSS.

3. Results

3.1. Soil variability in the study area

Soils and soil surface characteristics greatly differed between different landscape positions and vegetation types (Table 1). The steep-slope rangeland soils were the most degraded and showed the lowest values of total and particulate OM (OM, POM), OM content in different aggregate sizes: OM_{Agg}, $OM_{>250}$, $OM_{53\text{-}250}$, as well as in the ratio $OM_{>250}$/OM and POM/OM (Table 2). The fraction retained on different sieve sizes, F > 250, F53-250 and F > 53d, the aggregate stability in subsurface layer, total porosity and hydrophobicity were also lower than those of gentle-slope rangeland soils. Even though clay contents were low over the study area, the rangeland soil on the steep slope had higher clay contents and lower contents of coarse and very coarse sand, in comparison to the rangeland soil on the gentle slope. Although the differences were not significant, the steep slope rangeland soil also tended to present higher pH values and lower NaF pH values ($P < 0.1$) (Table 2).

POM, pH NaF, hydrofobicity and EC were identified by the discriminant analysis as the key variables to differentiate among the soils of different landscape positions and vegetation types. The model created five functions that allowed a good differentiation of groups ($p < 0.001$), and 89% of the soil samples were well classified. The first two discriminant functions explained 90% of the total variance. Fig. 2 shows the relationship between variables and plots in the canonical-axis space.

Axis 1, positively associated with POM, pH NaF and hidrofobicity, explained 56% of the total variance. According to the first discriminant function standardised by the common covariances, POM, which represents the labile OM, was the most important variable for discrimination on this axis. Soil samples from gentle-slope plantations and from the young plantation on the steep slope grouped towards positive values of axis 1, associated with higher values of POM and pH NaF. In the opposite position of the axis, both the rangeland and the mature plantation on the steep slope, and most samples from the gentle rangeland were negatively related to axis 1, with the rangeland plots on the steep slope showing the lowest values of POM and pH NaF. Mature afforestation plots were negatively related to axis 2, with higher values of EC and hydrophobicity.

3.2. Soil changes associated with afforestation

While rangeland soil was partially bare, the plantation soil was mainly covered by non-decomposed litter of pine needles. Beneath the fresh litter layer, an organic horizon in varying stages of decomposition developed (Table 1).

In the steep-slope sector, OM significantly increased in the plantations in comparison to the rangeland. However, in the gentle slope, OM increment was only found in the young plantation (Table 3; Fig. 3). OM associated with water-stable aggregates (OM_{Agg}) and particulate OM (POM) significantly increased ($P < 0.05$) in steep-slope plantations, relative to the rangeland (Fig. 3a). Although OM increased in the different fractions, the largest changes corresponded to the OM associated with large microaggregates ($OM_{53\text{-}250}$), which tripled in the young plantation in comparison to the rangeland (Table 3). In the gentle slope, despite the data variability, POM, OM_{Agg}, $OM_{>250}$ and $OM_{53\text{-}250}$ significantly increased only in the young plantation (Fig. 3b, Table 3).

The distribution of OM fractions varied among the study sites on the steep slope (Fig. 4a). The proportion of the most labile fractions significantly increased in steep-slope plantations in comparison with the rangeland. Both the proportion of OM associated with water-stable aggregates ($OM_{>250}$ and $OM_{53\text{-}250}$) and the proportion of POM significantly increased in the plantations. In contrast, the proportion of the recalcitrant fraction ($OM_{<53}$/OM), for both with and without chemical dispersion samples, significantly decreased in the afforestation, relative to the rangeland soils. On the other hand, the recalcitrant fraction in chemical dispersed soil samples increased only in the rangeland and the

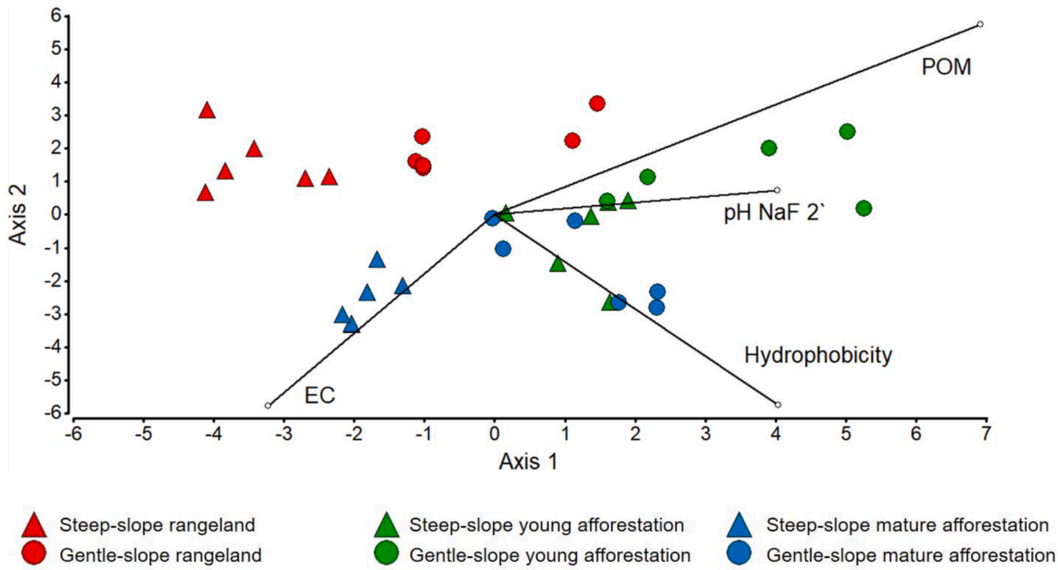

Fig. 2. Discriminant analysis biplot showing soil variables (vectors) and study sites.

Table 3
Soil properties of the study sites.

		OM[a]	$OM^{a}_{>250}$	$OM_{53\text{-}250}$[a]	pH	EC [b]	pH NaF 2‘	pH NaF 60‘	BD[c]	Porosity (%)	Hydrophobicity[d]	Aggregate stability[e] surface	Aggregate stability[e] subsurface
Steep slope	Rangeland	3.32 ± 0.22**a**	2.21 ± 0.38**a**	2.60 ± 0.37**a**	6.21 ± 0.06	59.10 ± 8.41**a**	7.94 ± 0.05**a**	8.71 ± 0.05**b**	1.23 ± 0.04	53.53 ± 1.41	8.5**a**(3–13)	2**a**(2–2)	5**a**(5–5)
	Young afforestation	6.62 ± 1.14**b**	4.29 ± 0.83**ab**	9.50 ± 1.82**b**	6.20 ± 0.07	59.24 ± 13.63**a**	8.48 ± 0.15**b**	9.22 ± 0.11**c**	1.02 ± 0.05	61.33 ± 2.02	24**b**(24–36)	6**b**(6–6)	6**b**(6–6)
	Mature afforestation	6.96 ± 1.05**b**	5.45 ± 0.94**b**	5.19 ± 0.54**a**	6.38 ± 0.06	122.23 ± 16.18**b**	7.72 ± 0.03**a**	8.40 ± 0.06**a**	1.10 ± 0.09	58.31 ± 3.53	24**b**(24–24)	6**b**(6–6)	6**b**(4–6)
Gentle slope	Rangeland	7.48 ± 1.13**a**	5.73 ± 1.39**a**	8.29 ± 1.62**a**	6.00 ± 0.07	79.23 ± 11.04	8.53 ± 0.23	9.05 ± 0.16	0.97 ± 0.05	63.36 ± 2.02	13**a**(8.5–13)	0**a**(0–0)	6(6–6)
	Young afforestation	16.54 ± 3.47**b**	13.81 ± 4.40**b**	18.43 ± 2.99**b**	6.06 ± 0.13	87.48 ± 21.58	8.09 ± 0.20	8.63 ± 0.27	0.83 ± 0.01	68.51 ± 0.25	24**b**(24–36)	6**b**(6–6)	6(6–6)
	Mature afforestation	5.78 ± 0.47**a**	5.06 ± 0.34**a**	6.73 ± 1.39**a**	6.28 ± 0.05	53.40 ± 4.56	8.33 ± 0.15	8.92 ± 0.16	0.86 ± 0.05	67.58 ± 1.83	36**b**(24–36)	6**b**(6–6)	6(6–6)

Different letters indicate soil variables that significantly differ between vegetation types of the same slope position.
[a]Organic matter contents (%): OM: total organic matter; $OM_{>250}$: OM associated with stable macroaggregates; $OM_{53\text{-}250}$: OM associated with stable large micro-aggregates.
[b]Electrical conductivity ($\mu S\ m^{-1}$).
[c]BD = bulk density ($g\ cm^{-3}$).
[d]Hydrophobicity according to “Molarity of an Ethanol Droplet” technique: 0%=Very hydrophilic; 3%= Hydrophilic; 5%= Slightly hydrophobic; 8.5%= Moderately hydrophobic; 13%= Strongly hydrophobic; 24%= Very strongly hydrophobic; 36%= Extremely hydrophobic (Doerr 1998).
[e]Aggregate stability classes range from 0 = Soil too instable to measure; to 6 = Strong stability (Herrick et al., 2001).

mature afforestation.

The results for the gentle slope were different. No significant changes in the proportion of POM and OM_{Agg} were found in the plantations in comparison to the rangeland (Fig. 4b). However, a significant rise in the $OM_{>250}$ fraction was found for the mature plantation.

Afforestation soils from the steep slope showed a higher proportion of water-stable aggregates ($F > 53$), mainly associated with an increment in stable macroaggregates ($F > 250$), as compared to the rangeland soils (Fig. 5a). A significant increase in stable macroaggregates was also found for the mature afforestation on the gentle slope (Fig. 5b).

The fraction < 53 μm in the rangeland and the mature afforestation soils on the steep slopes varied according to soil dispersion (Fig. 6a). The chemical dispersion implied the aggregate breakdown, evidenced by a significant increase in the fraction < 53 μm (i.e., $F < 53 < Fd < 53$). On the other hand, soils from the young afforestation on the steep slope and all the studied soils from the gentle slope showed no significant changes with the chemical dispersion (Fig. 6). However, when complete soil dispersion was applied (OM removal + chemical dispersion + ultrasound), the fraction < 53 μm significantly increased for all the study site soils (Fig. 6).

The difference between the fraction lower than 53 μm after complete dispersion ($Fcd < 53$), in comparison to $F < 53$ and $Fd < 53$, was significantly greater in plantations than in the rangeland for sites located on the steep slope ($p < 0.05$) (Fig. 6a). On the contrary, non-significant

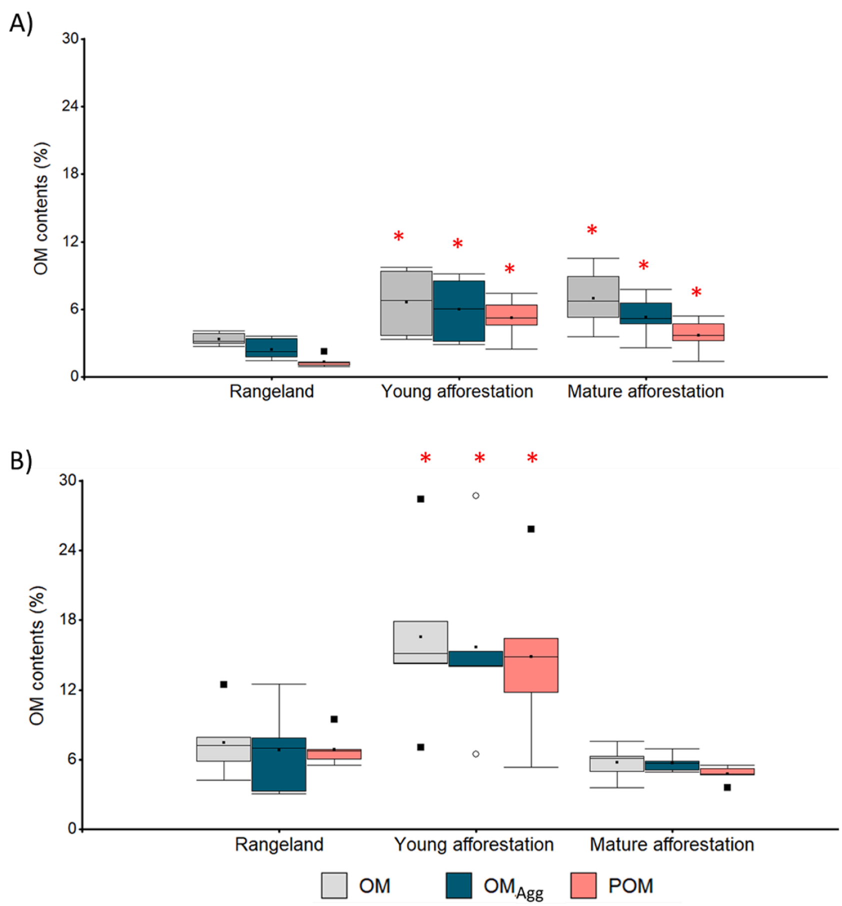

Fig. 3. Total organic matter (OM), organic matter associated with stable aggregates (OM_{Agg}) and particulate organic matter (POM) for the study sites on the steep (A) and the gentle (B) slopes.

differences were found among the study sites on the gentle slope.

Besides the changes found in OM fractions on the steep slope, we recorded higher pH NaF values in the young plantation and higher EC in the mature one, in comparison to rangelands (Table 3). Aggregate stability in the upper soil layer significantly increased in the plantations, relative to the rangelands. Although in the subsurface layer a significant increase in plantations was recorded for the steep slope, both gentle and steep slope rangelands tended to show strong subsurface aggregate stability (Table 3). Higher hydrophobicity was also recorded in plantations than in rangelands. While soils varied between hydrophilic and strongly hydrophobic in the rangelands, under afforestation, soils were always very strongly or extremely hydrophobic (Table 3).

3.3. *Soil erosion and runoff*

Soil erosion in the rangelands was higher on the steep than on the gentle slope (Fig. 7a). On the steep slope, significant differences between the treatments were found ($F = 15.84$, $p < 0.001$). Soil erosion significantly decreased in plantations compared to rangeland, even if fresh litter and organic layer were completely removed (Fig. 7a).

In contrast, on the gentle slope, soil erosion significantly decreased in the plantations in comparison to the rangeland only if the soil remained covered. When fresh litter and the organic layer were removed, erosion rates approached the values found in the rangeland (Fig. 7a).

Regarding runoff, ANOVA showed no interaction between sector and the rain treatment factors ($F = 1.03$, $P = 0.406$). Runoff in the young plantations was significantly higher than in rangelands, while the mature plantations showed intermediate values. In turn, runoff was significantly higher on steep than on gentle slopes (Table 4).

3.4. *Organic matter loss and enrichment ratios*

OM losses did not follow the same pattern as erosion rates. Although sediment yield in organic layer-removed plots was significantly lower in the steep slope plantations than in the rangeland (Fig. 7a), the OM losses were similar to those found in the rangeland (Fig. 7b). Even the young plantation without cover alteration, with very low erosion rates, showed OM losses close to those registered in the rangeland. On the gentle slope, the erosion rates in non-altered plantations were so low that there was not enough sediment for OM analysis. OM loss in the mature plantation was considered 0, since three of the four plots generated no sediment (sediment yield = 0 gm^{-2}). The maximum losses of OM were recorded in the young plantation where litter and the organic layer had been removed (Fig. 7b). The OM loss was positively and significantly correlated ($r = 0.38$; $P < 0.05$) with the difference between the fraction < 53 μm after complete dispersion (Fcd < 53) and without dispersion (F < 53).

The enrichment ratio showed values close to 1 in the rangelands, increasing in the plantations. On the steep slope, the enrichment ratio reached a maximum in the young afforestation, where OM content of sediments was more than five times as high as soil OM. On the gentle-slope plantations, the fresh litter and organic layer-removed plots also showed a significantly higher enrichment ratio than the rangeland, and OM content of sediments was at least twice as high as OM soil content. The enrichment ratio was positively correlated with the pH NaF60′ ($r = 0.49$; $P < 0.001$).

3.5. *Sediment texture*

In the plantations, sediments were significantly enriched in fine

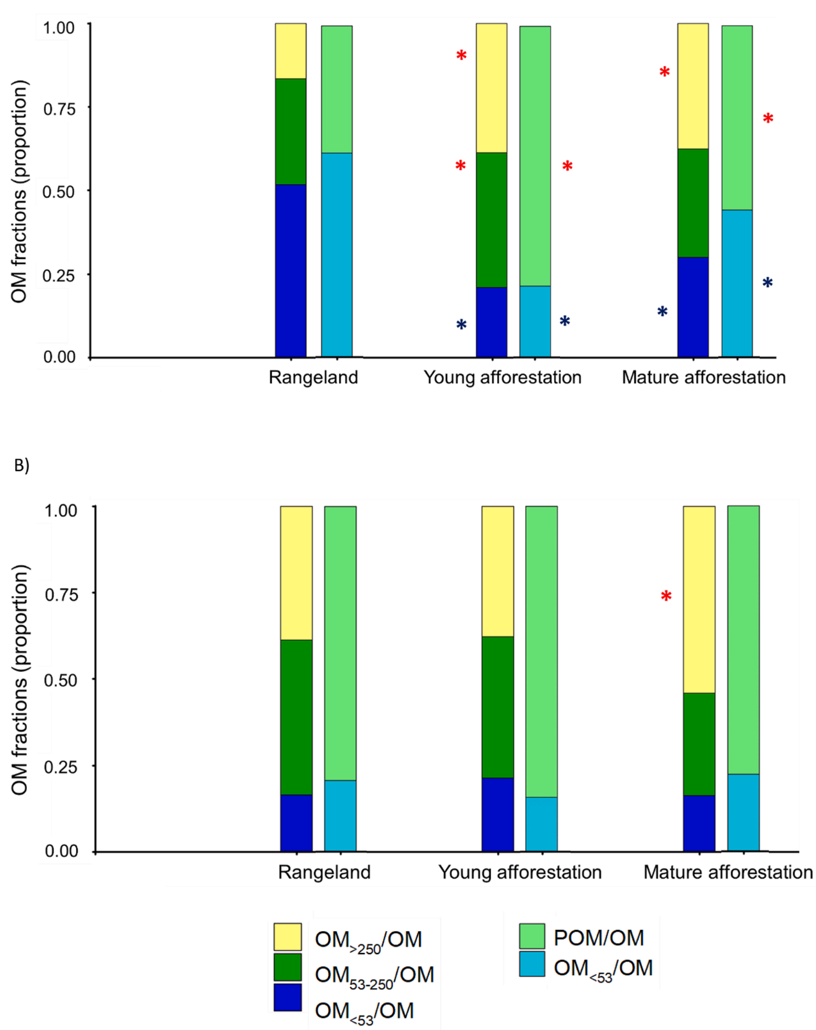

Fig. 4. Proportion of the different organic matter fractions for the study sites on the steep (A) and the gentle (B) slopes. The first column for each site refers to organic matter (OM) associated with water stable aggregates, including: OM associated with fraction>250 μm ($OM_{>250}$/OM), OM associated with 53–250 μm fraction ($OM_{53\text{-}250}$/OM) and OM associated with clay + silt fraction ($OM_{<53}$/OM). The second column for each site refers to the procedure with previous chemical dispersion, including: particulate OM fraction (POM/OM) and OM associated with clay + silt fraction ($OM_{<53}$/OM).

fractions, silt and/or clay, and impoverished in fine and medium sands (Fig. 8) in comparison to the original soil. The differential removal of the fine fraction was found in the four plantations, regardless of age or slope. For example, Fig. 9b shows the grain-size distribution in soil and sediments in a sampling point from the young plantation on the steep slope. A significant correlation between silt in sediments and OM enrichment ratio was found ($r = 0.39$, $P = 0.03$): the higher the enrichment in silt, the higher the OM enrichment rate.

On the contrary, sediments from rangelands were not enriched in fine fractions, but in coarse ones. Textures in soil and sediment were more variable in the rangelands, and for most size fractions no significant differences were found (Fig. 8). However, for samples showing selective removal, sediments were enriched in sand fraction (Fig. 9a).

4. Discussion

4.1. Soil changes associated with afforestation

4.1.1. Organic matter fractions

Although the coarse textures and the lack of non-crystalline clays dominated in the studied hillslope, the slope position implied a strong variation in soil properties, with soils on the gentle slope having the highest topsoil fertility, according to the analysed soil variables. In this study, soil fertility was mainly equated to soil organic matter content, since this variable is considered to be a good indicator of soil quality due to its influence on key functional properties, such as nutrient availability, soil structure and water relations (Ogle and Paustian, 2005). Differences found between the two slope gradients could be associated with erosion processes in the long term. In fact, the high bare soil cover (Table 1) and the depletion in coarse and very coarse sands in rangeland soils on the steep slope (Table 2) suggest active erosion processes since volcanic sand can be considered a light and erodible fraction (McDaniel et al., 2012) (see Fig. 8b, 9a).

These differences imply that the afforestation on the steep slope was installed on less fertile soils than those on the gentle one. However, certain changes in the soil associated with land-use change (from rangeland to afforestation), such as the increase in surface aggregate stability and the water repellency (i.e., hydrophobicity) occurred independently of the previous soil characteristics and the age of the plantation (Table 3).

On the other hand, changes in organic matter fractions were highly dependent on the initial OM content and the afforestation age. On the steep slope, both the young and mature plantations favoured an increment in total OM and in labile OM fractions, in comparison to the rangeland (Fig. 3a), and OM did not increase with afforestation age (Table 3). While soil OM was dominated by recalcitrant OM ($OM_{<53}$/OM), representing more than 50% in the rangeland, the plantation soil was dominated by the labile fractions (Fig. 4a). In the young afforestation, the fraction that increased the most was the OM associated with large microaggregates (i.e., $OM_{53\text{-}250}$), with three-fold values compared to rangeland soils, while the stable macroaggregates also showed a rise in OM content ($OM_{>250}$), with values twice as high as those found in

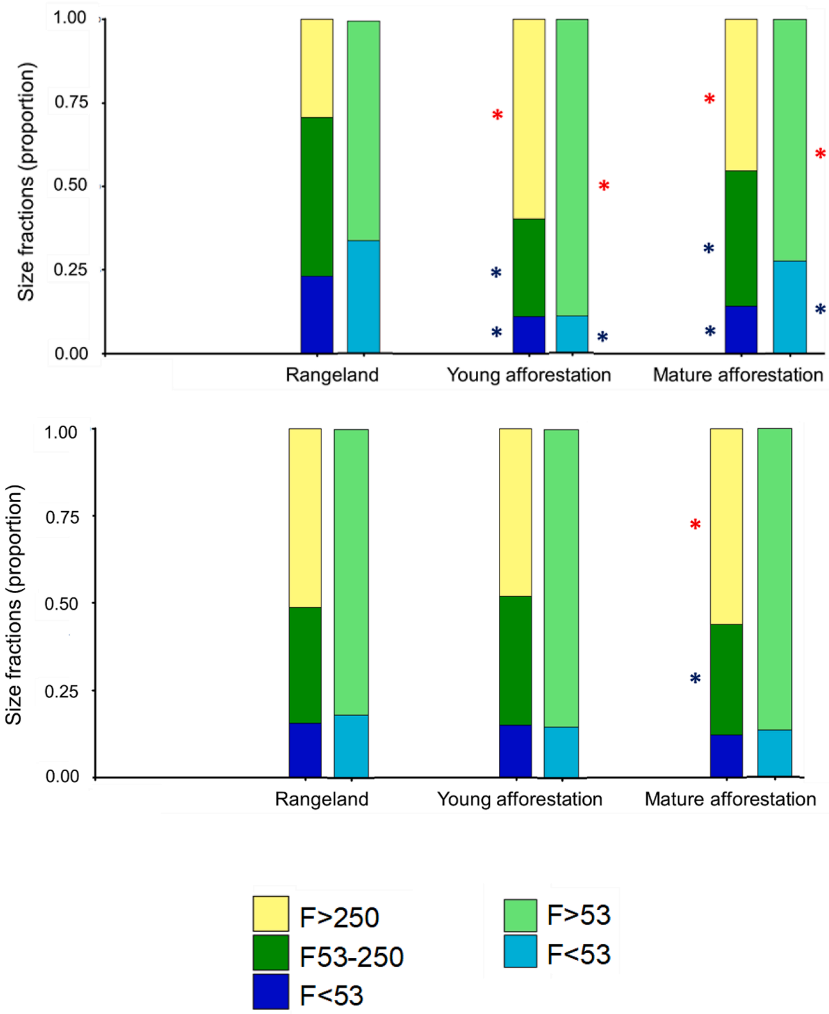

Fig. 5. Proportion of size fractions for the study sites on the steep (A) and the gentle (B) slopes. The first column for each site refers to water stable aggregates, including the proportion of macroaggregates ($F > 250$), microaggregates (F53-250) and clay + silt fraction ($F < 53$). The second column for each site refers to the procedure with previous chemical dispersion, including the fraction greater ($Fd > 53$) and lower ($Fd < 53$) than 53 μm.

rangeland soils (Table 3).

On the gentle slope, with OM contents in the rangelands twice as large as those found on the steep slope (Table 2), OM changes under pine plantation were not so evident. Labile OM was the dominant fraction for all the sites, even for rangeland soils (Fig. 4b). Total and labile OM were significantly higher than in the rangeland only in the young plantation, decreasing in the mature one (Fig. 3b, Table 3).

In xeric conditions, as in the study area, non-crystalline mineral formation is restricted, and halloysite formation, a 1:1 layer silicate, is expected, conditioned by precipitation and Si concentration (Parfitt et al., 1984). However, the absence of non-crystalline minerals in the soil surface could also be related to the intense use that these lands have suffered for more than a century. Overgrazing, mainly when plant cover is lost, might result in desiccation of soil, allowing non-crystalline materials to evolve to halloysite (Parfitt and Wilson, 1985). Soil mineralogy conditions OM storage capacity and stabilisation in volcanic soils (Feller and Beare, 1997; Kondo et al., 2010; Calabi-Floody et al., 2011; Hernández et al., 2012; Matus et al., 2014). The higher OM values found in soils on the lower slope relative to the upper slope could be influenced by the presence of imogolite and imogolite-halloysite transition in some lower-slope soil samples, according to pH NaF values (Irisarri, 2000) (Fig. 2, Table 3).

4.1.2. *Influence of afforestation age on soil organic matter*

Since plant inputs drive decomposition and soil OM formation (Paul, 2016), changes found in OM are conditioned by decomposition rates, which are specially modified when natural vegetation is replaced by exotic coniferous plantations (Araujo and Austin, 2015; González-Polo et al., 2019). Low decomposition rates of exotic conifer litter (Mazzarino et al., 1998) might hinder OM incorporation into mineral soil; in fact, a thick layer of non-decomposed pine needles is common, mainly in closed-canopy plantations (see Table 1). The recalcitrance of plant litter and rhizo deposits is mainly relevant during the initial phases of litter decomposition, which in some instances can last several years (Lützow et al., 2006). In this sense, the contribution of the undergrowth, herbaceous and native shrubs, which provide OM less rich in recalcitrant compounds than the pines, could favour OM enrichment in the young afforestation. Besides, the decomposition of pine litter is strongly influenced by incident solar radiation (Araujo and Austin, 2015). Thus, crown closure in the mature afforestation could condition the incorporation of OM into the mineral fraction. Studies in Ponderosa pine plantations in Patagonia showed that soil OM increased in thinned plantations, where understorey vegetation and incident solar radiation increased, in comparison to unthinned ones (Candan and Broquen, 2009). The increment in hydrophobicity found in the studied pine plantations (Table 3) could also restrict decomposition rates, since hydrophobicity reduces surface wettability, affecting the accessibility of

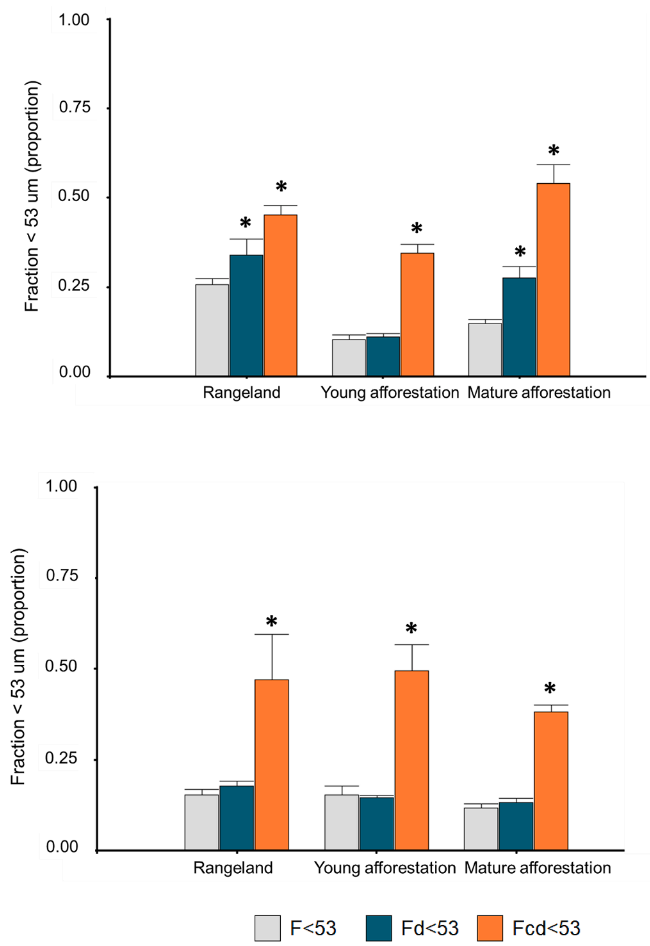

Fig. 6. Proportion of fraction lower than 53 μm for soil samples wet sieved, without (F < 53) and with (Fd < 53) previous chemical dispersion, and for samples completely dispersed (i.e, OM removal, chemical dispersed and ultrasounded) (Fcd < 53). Soils on the steep (A) and the gentle (B) slopes are shown.

microorganisms to OM and even their living conditions (Lützow et al., 2006).

The fact that soil OM does not increase with afforestation age, even though the time elapsed since afforestation was rather short, recalls the proposal of Six et al. (2002b), suggesting a soil C-saturation capacity, or a maximum soil C storage potential, determined by the physicochemical properties of the soil. In fact, native shrub and tree patches in the area, which have been preserved as forest relicts, had similar values to those found in the low rangelands (7.6%, Vogel pers. com.). This value was also similar to soil OM contents on the steep slope, 24 years after pine planting (6.96%). Coarse soil textures, as found in this study, may especially limit OM protection and stabilisation processes (Plaza et al. 2018), conditioning OM storage capacity. Since non-crystalline aluminosilicates are essential for determining soil carbon storage capacity in volcanic soils (Matus et al., 2014), their absence in most of the studied soils could also straighten OM storage.

4.1.3. Soil aggregates

Although the magnitude of the changes varied according to the initial soil OM contents and afforestation age, the results show that the greatest enrichment in OM in plantations, in comparison to rangelands, was associated with the 53–250 μm fraction, i.e., OM associated with large microaggregates, and with the fraction greater than 250 μm, i.e., OM associated with macroaggregates (Fig. 4). However, the formation of stable macroaggregates was enhanced in plantations (F > 250) (Fig. 5), and the proportion of OM associated with this fraction (OM > 250) was the most increased in comparison to rangeland soils. Although the methodologies used in our study were quite different, which makes the comparison difficult, our results agree with those of other studies on OM fractions. Studies in afforested and forested ecosystems found that the highest amount of carbon was stabilised within the microaggregates, both 53–250 μm microaggregates and microaggregates occluded within 250–2000 μm macroaggregates (Six et al. 2002a). Studies in sandy loam soils under natural forests showed that soil aggregation and carbon content were mainly related to fine POM (Rienzi et al., 2018).

Afforestation soils showed an enhancement in aggregate stability in surface soils (Table 3) and a higher proportion of water-stable macroaggregates (F > 250) than rangeland soils (Fig. 5). Other studies have shown higher aggregation in afforested ecosystems (Six et al., 2002a; Rienzi et al., 2018) and carbon accumulation in macroaggregates in forested plots that replaced crops (Jiang et al., 2019). This fraction exerts a minimal amount of physical protection (Elliot, 1986) and corresponds to the biochemically protected OM, i.e., the stabilisation of OM due to its own chemical composition (i.e., recalcitrant compounds such as lignin and polyphenols) and through chemical complexing processes in soil (Six et al., 2002b). Recent studies have shown that when inputs of OM are added to the soil, the new C is accumulated more but decomposed faster in macroaggregates than in microaggregates (Peng et al., 2017). A current view on soil OM, which considers soil as a system with functional complexity, proposes that decomposition and organic carbon protection are controlled by the molecular diversity of the organic compounds, the soil spatial heterogeneity and the temporal variations in the soil environment (Lehmann et al. 2020).

As expected, the chemical dispersion implied an increase in the fraction <53 μm, by aggregate breakdown. However, this rise was noted only in the rangelands and the mature plantations, with significant changes on the steep slope, and only a tendency on the gentle slope (Fig. 6). In contrast, soils from young plantations showed no change in F < 53 with the chemical dispersion. The lack of aggregate dispersion in soils from the young plantations and the low increase in Fd < 53 in soils from the gentle slope (Fig. 6b) suggest, a priori, that in these soils chemical dispersion hardly generates aggregate breakdown. However, a dramatic increase in the release of <53 μm particles was found with complete dispersion (i.e. OM removal + chemical dispersion + ultrasound) (Fig. 6). These results suggest that the studied soils, despite their coarse texture and absence of allophane, form very stable microaggregates, formed by OM bound to clay + silt fractions, which are only broken when the cementing agents are completely removed. Oades (1984) was the first to theorise the formation of microaggregates within macroaggregates, and later studies on aggregate formation extensively corroborated this theory (Six et al., 2004). Agents stabilising microaggregates are not only OM compounds but also cementing agents, such as oxides, hydroxides, oxyhydroxides (Oades and Waters, 1991) and polysaccharides (Six et al., 2004). These stable microaggregates, in turn, are bound together into macroaggregates (>250 μm) by temporary binding agents such as roots and hyphae and by transient binding agents such as microbial- and plant-derived polysaccharides and proteins (Six et al., 2004; Totsche et al., 2018). The role of OM in aggregate stability is especially important in variably charged soils (Yu et al., 2020), such as volcanic soils. Recent studies showed that OM stabilisation in soil aggregates is highly influenced by soil bacterial diversity, which is affected by afforestation (Zhao et al., 2018). The formation and degradation of microaggregates was shown to be more dynamic than previously thought (Jastrow et al., 1996). Studies on mean residence time of OM showed that the light fraction, comprising intra-aggregate plant residues, and the inter-aggregate POM have short mean residence times of about 3 and 12 years, respectively (Paul, 2016). In contrast, studies on surface layers of forest volcanic soils found mean residence times of the light fraction ranging between 6 and 150 years (Kondo et al., 2010).

Recent studies reported apparent lack of aggregate hierarchy in Andisoles, due to the lack of size-dependent change in carbon concentration after weak dispersion treatments (Hoyos and Comerford, 2005;

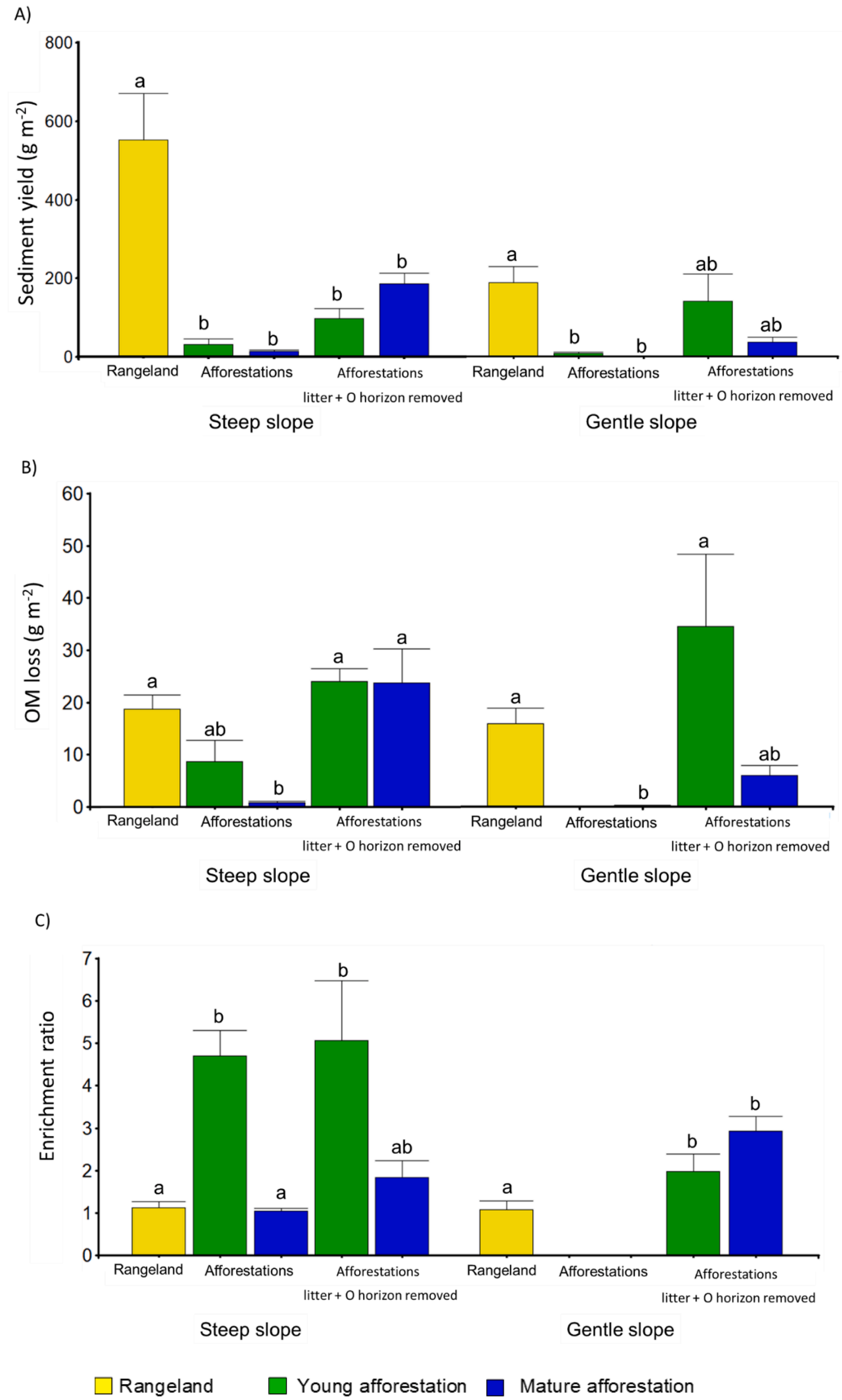

Fig. 7. Mean sediment production (A), organic matter lost by erosion (B) and organic matter enrichment ratio in sediments (C) in rainfall simulation experiments developed in rangelands, in plantations with unmodified soil and in plantations with removal of fresh litter and organic horizon.

Table 4
Means and standard errors of runoff rate (mm h^{-1}) for rainfall simulation experiments.

	Rangeland **a**	Young afforestation **c**	Mature Afforestation **a**	Young afforestation duff-removed **bc**	Mature Afforestation duff-removed **ab**	F = 5.02 (p = 0.003)
Steep slope **a**	27.8 ± 2.8	47.5 ± 3.2	19.3 ± 10.9	33.5 ± 2.3	25.2 ± 5.7	
Gentle slope **b**	7.5 ± 0.6	32.1 ± 8.8	17.8 ± 10.1	34.6 ± 4.4	18.0 ± 4.4	
F = 4.71 (p = 0.038)						

Different letters indicate significant differences between sectors (rows) and between rain treatments (columns).

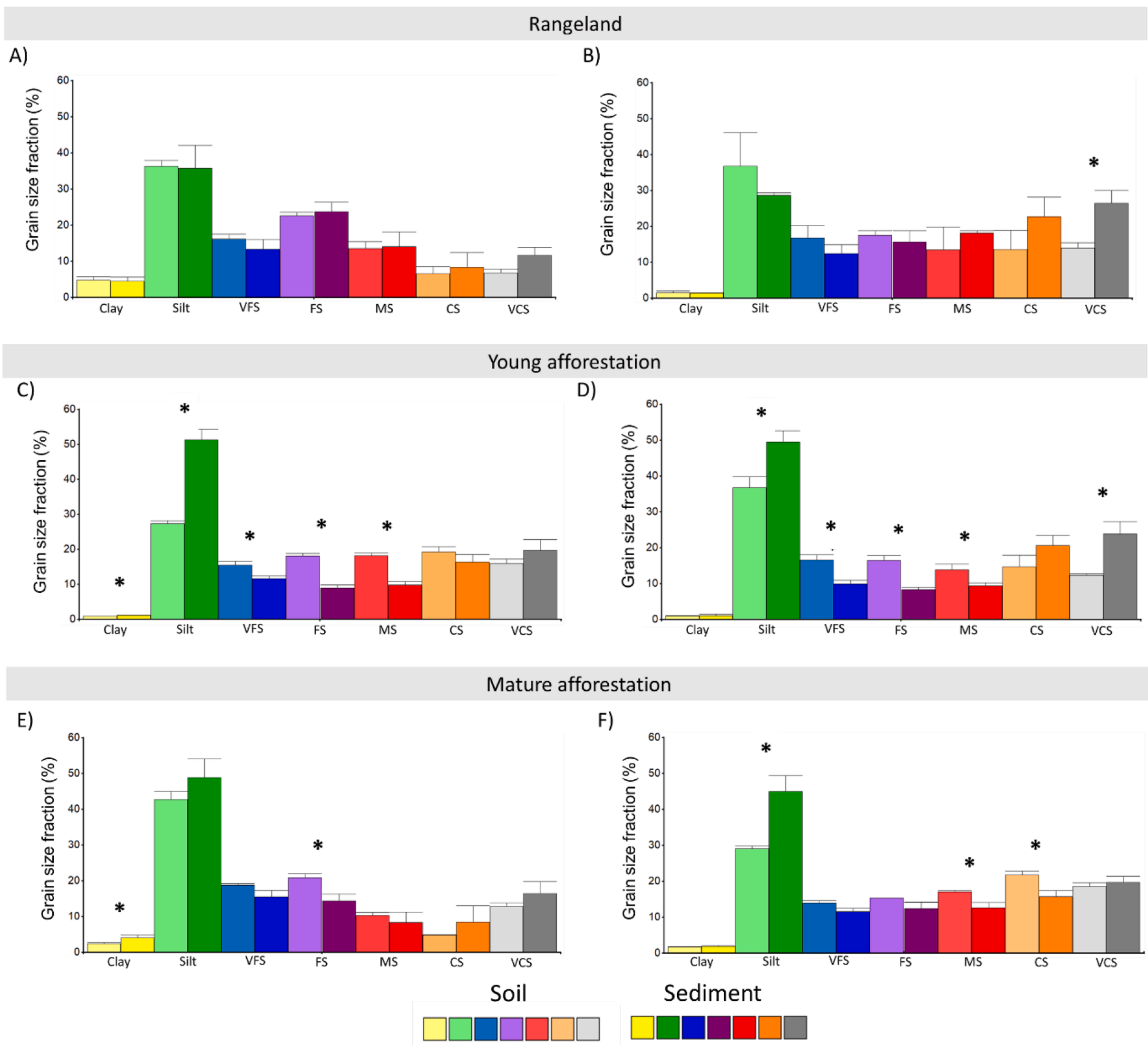

Fig. 8. Gran size fractions of soils and sediments for the study sites on the steep (A, C, E) and the gentle (B, D, F) slopes. VFS: very find sand; FS: fine sand; MS: mean sand; CS: coarse sand; VCS: very coarse sand. The size fraction were determined by laser diffraction, expect VCS, which was separated by sieving.

Candan and Broquen, 2009). However, Asano and Wagai (2014) showed that the aggregate hierarchy in Andisols is present, but at smaller spatial scales, and maintained by stronger forces due to higher concentrations of OM and non-crystalline clays, in comparison to non-volcanic soils. Despite the absence of allophane in our soils, the dramatic increase in Fcd < 53 found (Fig. 6) agrees with Asano and Wagai (2014), since the main criterion to identify aggregate hierarchy is the aggregate breakdown and concurrent release of silt- and clay-sized particles only after increased levels of dispersion energy (Oades and Waters, 1991).

The differences between Fcd < 53, in comparison to F < 53 and Fd < 53, representing microaggregates occluded within macroaggregates, were significantly greater in plantations than in the rangeland for the steep-slope sites (Fig. 6a). These results suggest that plantations not only increase POM but significantly increase the formation of microaggregates.

4.2. *Erosion processes*

4.2.1. *Erosion and runoff rates*

Potential erosion rates found in this study agree with similar simulated rainfall assays conducted on different types of soils from the Andean Patagonian region (La Manna et al., 2016). The erodibility of these

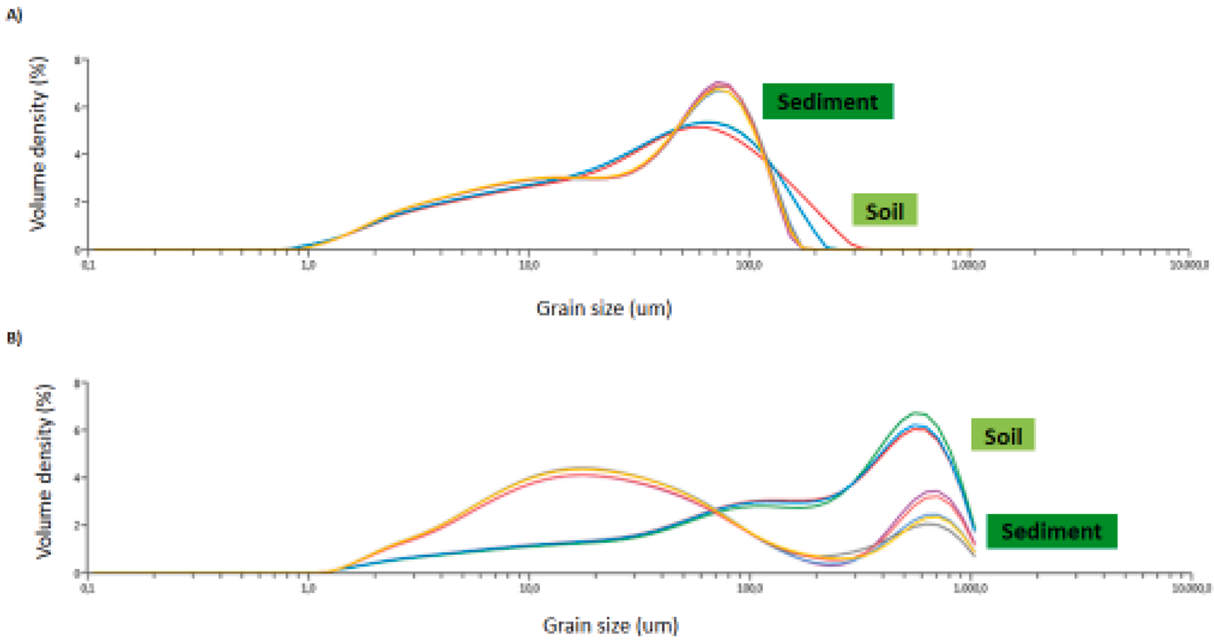

Fig. 9. Grain size distribution for soil and sediment samples from the rangeland on the gentle slope (A) and the young afforestation on the steep slope (B).

non-allophanic volcanic soils, both in the rangeland and plantations where fresh litter and organic layers were removed, resulted in even higher values than those recorded in steppe degraded soils with very low OM contents (Rostagno, 1989). Studies on Andisols have shown their high susceptibility to wind and water erosion when surface cover is removed or degraded (Dahlgren et al., 2004; Mc Daniels et al., 2012; Morales et al., 2013), and volcanic soils without non-crystalline clays would be the most susceptible ones (La Manna et al., 2016).

In steep-slope plantations, erosion rates significantly decreased, in comparison to the rangeland, even where fresh litter and organic layers were completely removed (Fig. 7a). This result could be associated with both the very high erodibility value found in the rangeland and the positive changes registered in the mineral soil, which could generate greater resistance to erosion.

In contrast, on the gentle slope, where rangeland soils are more fertile and only slight changes were found in the afforested soil, the potential erosion significantly decreased only when the soil remained covered. When the soil cover was removed, potential erosion rates approached the rates found in the rangeland (Fig. 7a).

Although runoff was highly variable, it was greater on the steep slope, as expected, due to the effect of the slope on runoff and to the greater soil porosity of gentle-slope soils, which would favour infiltration. The maximum values were recorded in the young plantations, regardless of whether litter and organic horizon were removed or not, while the mature plantations exhibited intermediate values (Table 4). The increase in runoff in the plantations could be explained by the significant increment in water repellency (Table 3), associated with hydrophobic compounds leached from pine litter (Doerr et al., 1998; Jaramillo Jaramillo, 2005; Maia et al., 2010). The high organic values found in the young plantations (Fig. 3) could explain the maximum values of runoff, since water repellency increases in severity with increasing soil organic carbon contents (Harper et al., 2000; Jaramillo et al., 2000).

4.2.2. Relations between organic matter, soil aggregates and erosion processes

OM losses do not follow the same pattern as erosion rates. Despite the effects of plantations on erosion rates (Fig. 7a), OM losses in most situations were as high in plantations as in rangelands (Fig. 7b). These results could be explained by the high enrichment of sediments with organic matter in afforestation, i.e., lower amount of sediments but rich in organic matter. The OM enrichment ratio in sediments, which can be understood as a soil depletion rate, was significantly higher in plantations, in comparison to rangelands. While in rangelands values were close to 1, in the forested plots they were always higher (Fig. 7c). In part, these results could be related to potential erosion rates, since the enrichment ratio of OM decreased with increasing soil loss (Schiettecatte et al., 2008). The enrichment of sediments with organic carbon varies widely, influenced by soil properties, such as texture, aggregation, initial OM content, as well as by rainfall intensities and microtopography (Hu et al., 2013). Erosion/deposition processes imply carbon distribution patterns in surface soil, varying according to soil properties, land use and geomorphology (Boix-Fayos et al., 2009; Kirkels et al., 2014). In our study, the OM loss was positively correlated with the difference between the fraction lower than 53 μm after complete dispersion (Fcd < 53) and without or with weak dispersion (F < 53, Fd < 53), suggesting that OM loss by erosion is enhanced by the formation of soil microaggregates.

The OM enrichment of sediments from plantations was associated with enrichment in fine fractions (Figs. 8 and 9), in agreement with other studies (Costantini and Loch, 2002). Studies on the distribution of soil organic carbon within particle-size fractions of volcanic soils in Mexico showed that soil organic carbon was mainly associated with the silt-size fraction (Covaleda et al., 2011).

The enrichment in fine fractions found in sediments may be related to the formation and removal of soil aggregates. Thus, the maximum enrichment ratios were found in the young plantations on the steep slope, where OM content in sediments was up to five times as high as soil OM (Fig. 7c). This young afforestation showed the greatest changes in OM fractions (Fig. 4a, Table 3), and soils needed high dispersal forces to allow the breakdown of aggregates (Fig. 6a). Interestingly, some soil samples from this afforestation showed high pH NaF values, suggesting the presence of imogolite (Table 3). The presence of non-crystalline clays could also favour soil aggregation (Matus et al., 2014) and the interaction between fine particles and OM, increasing the OM enrichment ratio in sediments.

These results suggest that the improvements in OM, particularly in POM and OM associated with stable aggregates, and the formation of soil microaggregates favour the loss of OM when the soil undergoes erosion processes. These results agree with those of other studies on volcanic soils, suggesting that erosion processes involve the detachment of soil microaggregates, which could be mobilised with no previous dispersion (Poulenard et al., 2001; Rodríguez Rodríguez et al., 2002, 2006).

However, in rangelands with lower values of OM, OM_{Agg} and POM (Fig. 3) and low surface soil aggregate stability (Table 3), no selective removal of fine particles was found. On the contrary, erosion involved the loss of single particles, mainly coarse and very coarse sands (Fig. 8) or probable macroaggregates rich in fine sands (Fig. 9a). In fact, soils from the gentle-slope rangeland showed significantly higher contents of coarse and very coarse sands than the upper-slope rangeland soil. This result might explain why coarse-textured soils were the most erodible ones in rangelands developed on different soils (La Manna et al., 2016). It is well known that volcanic porous sands are considered a light fraction (Nanzyo and Kanno, 2018), susceptible to erosion.

Our results suggest a close and complex relationship between soil OM, aggregation, soil vulnerability to erosion, and OM losses in afforested volcanic soils, modifying OM dynamic. Although plantations in these coarse-textured soils were found to be effective in controlling erosion, with low potential erosion rates when the soil remained covered, even on steep slopes, the sediments were highly enriched in OM. The improvement in labile OM fractions and aggregate formation associated with plantations, mainly in the less fertile soils, may increase the risk of erosion of microaggregates rich in OM and fine textural fractions. Therefore, misguided forest management and logging strategies can involve large losses of carbon in these volcanic soils.

5. Conclusions

Changes in soil OM associated with afforestation greatly varied depending on the initial soil OM contents and the age of the plantation. Less fertile soils showed significant positive changes, with increments in OM, POM and OM associated with stable macro and microaggregates. The more fertile soils on the gentle slope also showed positive changes in OM associated with the afforestation, but these changes were more erratic and, mostly, of low magnitude. On the other hand, older plantations did not imply an increase in soil OM, at least in the first centimetres of the soil. These results, varying according to the age and initial OM content, agree with the great variation in results found in studies from Patagonia, which do not allow clear trends to be established regarding the increase in OM associated with coniferous plantations.

The rise in POM and microaggregation found in afforestations seemed to favour OM loss by erosion when the soils remained uncovered. Erosion in afforestation involved the removal of microaggregates rich in OM and silt fractions; in contrast, coarse and very coarse sand single particles were mainly lost in the rangelands.

Afforestation replacing degraded rangelands may control erosion in these highly erodible soils, as long as the soil remains covered. Otherwise, the loss of soil enriched in OM can favour soil carbon depletion.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

We acknowledge Pedro Noli, Marcos Antilef and Braian Vogel for their assistance during field and/or laboratory works. We appreciate A. Enriquez's suggestions in relation to laboratory techniques. This research was supported by the National Agency for Scientific and Technological Promotion (PICT-ANPCyT 1876-13).

References

Amiotti, N.M., Zalba, P., Sánchez, L.F., Peinemann, N., 2000. The impact of single trees on properties of loess-derived grasslands soils in Argentina. Ecology 81, 3283–3290. https://doi.org/10.1890/0012-9658(2000)081[3283:TIOSTO]2.0.CO;2.

Araujo, P.I., Austin, A.T., 2015. A shady business: pine afforestation alters the primary controls on litter decomposition along a precipitation gradient in Patagonia, Argentina. J. Ecol. 103(6), 1408-1420. https://doi: 10.1111/1365-2745.12433.

Asano, M., Wagai, R., 2014. Evidence of aggregate hierarchy at micro- to submicron scales in an allophanic Andisol. Geoderma 216, 62–74. https://doi.org/10.1016/j.geoderma.2013.10.005.

Avnimelech, Y., McHenry, J., 1984. Enrichment of transported sediments with organic carbon nutrients and clay. Soil Sci. Soc. Am. J. 48, 259–266. https://doi.org/10.2136/sssaj1984.03615995004800020006x.

Beeskow, A.M., Elissalde, N.O., Rostagno, C.M., 1995. Ecosystem change associated with grazing intensity on the Punta Ninfas rangelands of Patagonia, Argentina. J. Range. Manag. 48, 517–522.

Berthrong, S.T., Jobbágy, E.G., Jackson, R.B., 2009. A global meta-analysis of soil exchangeable cations, pH, carbon, and nitrogen with afforestation. Ecol. Appl. 19, 2228–2241. https://doi.org/10.1890/08-1730.1.

Błońska, E., Lasota, J., 2017. Soil organic matter accumulation and carbon fractions along a moisture gradient of forest soils. Forests 8, 448. https://doi.org/10.3390/f8110448.

Boix-Fayos, C., de Vente, J., Albaladejo, J., Martínez-Mena, M., 2009. Soil carbon erosion and stock as affected by land use changes at the catchment scale in Mediterranean ecosystems. Agric. Ecosyst. Environ. 133, 75–85. https://doi.org/10.1016/j.agee.2009.05.013.

Buduba, C.G., 2006. Modificaciones en el pH y contenido de materia orgánica en suelos del ecotono estepa/bosque andino patagónico por implantación de pino ponderosa. Dissertation. Universidad de Buenos Aires.

Buduba, C.G., Defossé, G.E., Irisarri, J.A., 2017. Impact of ponderosa pine afforestations on soil organic matter (SOM) in semiarid steppes of western Patagonia. Argentina. Agroforest. Syst. 91, 895–900. https://doi.org/10.1007/s10457-016-9963-6.

Calabi-Floody, M., Bendall, J.S., Jara, A.A., Welland, M.E., Theng, B.K., Rumpel, C., de la Luz Mora, M., 2011. Nanoclays from an Andisol: extraction, properties and carbon stabilization. Geoderma 161 (3–4), 159–167. https://doi.org/10.1016/j.geoderma.2010.12.013.

Cambardella, C.A., Elliott, E.T., 1993. Carbon and nitrogen distribution in aggregates from cultivated and native grassland soils. Soil. Sci. Soc. Am. J. 57, 1071–1076. https://doi.org/10.2136/sssaj1993.03615995005700040032x.

Cambardella, C.A., Gajda, A.M., Doran, J.W., Wienhold, B.J., Kettler, T.A., 2001. Estimation of particulate and total organic matter by weight loss-on-ignition. In: Lal, R., Kimble, J.M., Follett, F., Stewart, B.A. (Eds.), Assessment Methods for Soil Carbon. CRC Press, Boca Raton, Florida, pp. 349–359.

Candan, F., Broquen, P., 2009. Aggregate stability and related properties in NW Patagonian Andisols. Geoderma 154, 42–47. https://doi.org/10.1016/j.geoderma.2009.09.010.

Cao, L., Liang, Y., Wang, Y., Lu, H., 2015. Runoff and soil loss from *Pinus massoniana* forest in southern China after simulated rainfall. Catena. 129, 1–8. https://doi.org/10.1016/j.catena.2015.02.009.

Costantini, A., Loch, R.J., 2002. Effects of site preparation on runoff, erosion, and nutrient losses from *Pinus* plantations established on the coastal lowlands of south-east Queensland, Australia. Aust. J. Soil Res. 40 (8), 1287–1302. https://doi.org/10.1071/SR02032.

Covaleda, S.J., Gallardo, F., García-Oliva, F., Kirchmann, H., Prat, B., Etchevers, J.D., 2011. Land-use effects on the distribution of soil organic carbon within particle-size fractions of volcanic soils in the Transmexican Volcanic Belt (Mexico). Soil Use. Manag. 27 (2), 186–194. https://doi.org/10.1111/j.1475-2743.2011.00341.x.

Dahlgren, R.A., Saigusa, M., Ugolini, F.C., 2004. The nature, properties and management of volcanic soils. Advances in Agronomy.

Davis, M.R., Condron, L.M., 2002. Impact of grassland afforestation on soil carbon in New Zealand: a review of paired-site studies. Aust. J. Soil Res. 40 (4), 675–690. https://doi.org/10.1071/SR01074.

Di Rienzo, J.A., Casanoves, F., Balzarini, M.G., Gonzalez, L., Tablada, M., Robledo, C.W., 2013. InfoStat versión 2013. Universidad Nacional de Córdoba, Argentina infostat.com.ar.

Doerr, S.H., 1998. On standardizing the 'Water Drop Penetration Time' and the 'Molarity of an Ethanol Droplet' techniques to classify soil hydrophobicity: a case study using medium textured soils. Earth Surf. Proc. Land. 23, 663–668. https://doi.org/10.1002/(SICI)1096-9837(199807)23:7<663::AID-ESP909>3.0.CO;2-6.

Doerr, S., Shakesby, R., Walsh, R., 1998. Spatial variability of soil hydrophobicity in fire-prone *Eucalyptus* and Pine forests, Portugal. Soil Sci. 163, 313–324. https://doi.org/10.1097/00010694-199804000-00006.

Elliot, E.T., 1986. Aggregate structure and carbon, nitrogen and phosphorous in native and cultivated soils. Soil Sci. Soc. Am. J. 50, 627–633. https://doi.org/10.2136/sssaj1986.03615995005000030017x.

Enriquez, A.S., Cremona, M.V., 2018. Particulate organic carbon is a sensitive indicator of soil degradation related to overgrazing in Patagonian wet and mesic meadows. Wetl. Ecol. Manag. 26, 345–357. https://doi.org/10.1007/s11273-017-9577-4.

Feller, C., Beare, M.H., 1997. Physical control of soil organic matter dynamics in the tropics. Geoderma 79 (69–1), 16. https://doi.org/10.1016/S0016-7061(97)00039-6.

Fieldes, M.P., Perrot, K.W., 1966. The nature of allophane in soils. III. Rapid field and laboratory test for allophane. N. Z. J. Sci. 9, 623–629.

Gomez, F., Tarabini, M., Buduba, C.G., von Muller, A., La Manna, L., 2019. Balance hídrico en plantaciones de *Pinus radiata* en el NO de la Patagonia Argentina. Ecología Austral. 29 (2), 261–271. https://doi.org/10.25260/EA.19.29.2.0.776.

González-Polo, M., Bahamonde, H.A., Peri, P.L., Mazzarino, M.J., Fariña, C., Caballé, G., 2019. Soil microbial processes in a pine silvopastoral system in NW Patagonia. Agroforest. Syst. 93, 255–266. https://doi.org/10.1007/s10457-018-0210-1.

Gu, Z.J., Wu, X.X., Zhou, F., Luo, H., Shi, X.Z., Yu, D.S., 2013. Estimating the effect of *Pinus massoniana* lamb plots on soil and water conservation during rainfall events using vegetation fractional coverage. Catena. 109, 225–233. https://doi.org/10.1016/j.catena.2013.03.008.

Harper, R., Mckissock, I., Gilkes, R., Carter, D., Blackwell, P., 2000. A multivariate framework for interpreting the effects of soil properties, soil management and land use on water repellency. J. Hydrol. 231, 371–383. https://doi.org/10.1016/S0022-1694(00)00209-2.

Hernández, Z., Almendros, G., Carral, P., Álvarez, A., Knicker, H., Pérez-Trujillo, J.P., 2012. Influence of non-crystalline minerals in the total amount, resilience and molecular composition of the organic matter in volcanic ash soils (Tenerife Island, Spain). Eur. J. Soil Sci. 63 (5), 603–615. https://doi.org/10.1111/j.1365-2389.2012.01497.x.

Herrick, J.E., Whitford, W.G., De Soyza, A.G., Van Zee, J.W., Havstad, K.M., Seybold, C. A., Walton, M., 2001. Field soil aggregate stability kit for soil quality and rangeland health evaluations. Catena. 44 (1), 27–35. https://doi.org/10.1016/S0341-8162(00)00173-9.

Hoyos, N., Comerford, N.B., 2005. Land use and landscape effects on aggregate stability and total carbon of Andisols from the Colombian Andes. Geoderma 129 (3–4), 268–278. https://doi.org/10.1016/j.geoderma.2005.01.002.

Hu, Y., Fister, W., Kuhn, N.J., 2013. Temporal Variation of SOC Enrichment from Interrill Erosion over Prolonged Rainfall Simulations. Agriculture. 3, 726–740. https://doi.org/10.3390/agriculture3040726.

IRAM-SAGPyA, 29571-1., 2008. Calidad Ambiental-Calidad del suelo. Determinación de materia orgánica en suelos. Parte 1, Método de pérdida por calcinación.

Irisarri, J., 2000. La propuesta de reclasificación de los Andepts de Argentina, de acuerdo al Orden Andisoles. Workshop Soil Taxonomy. Instituto Nacional de Tecnología Agropecuaria, AICET, Asociación Argentina de la Ciencia del Suelo. 18–27.

Irisarri, J., Mendía, J., 1997. Relaciones suelo-paisaje en la evaluación de la potencialidad forestal de la región central andino— patagónica. Argentina. Bosque. 18, 21–30. https://doi.org/10.4206/bosque.1997.v18n1-03.

Jaramillo, D.F., Dekker, L.W., Ritsema, C.J., Hendrickx, J.M., 2000. Occurrence of soil water repellency in arid and humid climates. J. Hydrol. 231, 105–111. https://doi.org/10.1016/S0022-1694(00)00187-6.

Jaramillo Jaramillo, D.F., 2005. Humedad crítica y repelencia al agua en Andisoles colombianos bajo cobertura de *Pinus patula* schltdl y cham. Rev. Fac. Nal. Agr. Medellin. 58 (2), 2893–2906. https://doi.org/10.15446/rfnam.

Jastrow, J.D., Miller, R.M., Boutton, T.W., 1996. Carbon dynamics of aggregate-associated organic matter estimated by carbon-13 natural abundance. Soil Sci. Soc. Am. J. 60 (3), 801–807. https://doi.org/10.2136/sssaj1996.03615995006000030017x.

Jiang, R., Gunina, A., Qu, D., Kuzyakov, Y., Yu, Y., Hatano, R., Frimpong, K.A., Li, M., 2019. Afforestation of loess soils: Old and new organic carbon in aggregates and density fractions. Catena. 177, 49–56. https://doi.org/10.1016/j.catena.2019.02.002.

Kirkels, F.M.S.A., Cammeraat, L.H., Kuhn, N.J., 2014. The fate of soil organic carbon upon erosion, transport and deposition in agricultural landscapes — a review of different concepts. Geomorphology 226, 94–105. https://doi.org/10.1016/j.geomorph.2014.07.023.

Kondo, M., Uchida, M., Shibata, Y., 2010. Radiocarbon-based residence time estimates of soil organic carbon in a temperate forest: case study for the density fractionation for Japanese volcanic ash soil. Nucl. Instrum. Meth. B. 268 (7–8), 1073–1076. https://doi.org/10.1016/j.nimb.2009.10.101.

La Manna, L., Buduba, C.G., Rostagno, C.M., 2016. Soil erodibility and quality of volcanic soils as affected by pine plantations in degraded rangelands of NW Patagonia. Eur. J. For. Res. 135 (4), 643–655. https://doi.org/10.1007/s10342-016-0961-z.

La Manna, L., Gaspar, L., Tarabini, M., Quijano, L., Navas, A., 2019. ^{137}Cs inventories along a climatic gradient in volcanic soils of Patagonia: potential use for assessing medium term erosion processes. Catena. 181, 104089 https://doi.org/10.1016/j.catena.2019.104089.

La Manna, L., Tarabini, M., Gomez, F., 2018. ¿Son las plantaciones una herramienta para mitigar áreas degradadas? Una mirada desde el suelo. En: Massara Paletto V, Rostagno M, Buono G, Gonzalez C. & Ciano N. Restauración ecológica en la Diagonal Árida de la Argentina. INTA, IADIZA, UNPSJB, IPEEC-CONICET.

Laclau, P., 2003. Biomass and carbon sequestration of ponderosa pine plantations and native cypress forests in Northwest Patagonia. Forest Ecol. Manag. 180, 317–333. https://doi.org/10.1016/S0378-1127(02)00580-7.

Laclau, P., Andenmatten, E., Letourneau, F., Loguercio, G.A., 2008. Carbon Sequestration of Ponderosa Pine Plantations in Northwestern Patagonia. In: Bravo, F (eds.). Manag. For Ecosyst: The Challenge of Climate Change. 247-267. https://doi.org/10.1007/978-1-4020-8343-3_14.

Lehmann, J., Hansel, C.M., Kaiser, C., Kleber, M., Maher, K., Manzoni, S., Nunan, N., Reichstein, M., Schimel, J.P., Torn, M.S., Wieder, W.R., Kögel-Knabner, I., 2020. Persistence of soil organic carbon caused by functional complexity. Nat. Geosci. 13 (8), 1–6. https://doi.org/10.1038/s41561-020-0612-3.

Lima, A.M., Silva, I.R., Neves, J.C., Novais, R.F., Barros, N.F., Mendonça, E.S., Leite, F.P., 2006. Soil organic carbon dynamics following afforestation of degraded pastures with *eucalyptus* in southeastern Brazil. Forest Ecol. Manag. 235 (1–3), 219–231. https://doi.org/10.1016/j.foreco.2006.08.331.

Lu, G., Sakagami, K.I., Tanaka, H., Hamada, R., 1998. Role of soil organic matter in stabilization of water-stable aggregates in soils under different types of land use. Soil Sci. Plant Nutr. 44 (2), 147–155. https://doi.org/10.1080/00380768.1998.10414435.

Lützow, M., Kögel-Knaber, I., Ekschmitt, A., Matzner, E., Guggenberger, G., Marschner, B., Flessa, H., 2006. Stabilization of organic matter in temperate soils: mechanisms and their relevance under different soil conditions - a review. Eur. J. Soil Sci. 57 (4), 426–445. https://doi.org/10.1111/j.1365-2389.2006.00809.x.

Maia, C., Fukamachi, C., Cambronero, Y., Dedecek, R.A., Mangrich, A.S., Narimoto, K., Milori, D., Simões, M., 2010. Hydrophobicity of an Entisol under loblolly pine (*Pinus taeda*) plantation. Braz J. For. Res. 30 (10), 1–8. https://doi.org/10.4336/2010.pfb.30.62.93.

Mao, R., Zeng, D.H., 2010. Changes in soil particulate organic matter, microbial biomass, and activity following afforestation of marginal agricultural lands in a semi-arid area of Northeast China. Environ. Manage. 46 (1), 110–116. https://doi.org/10.1007/s00267-010-9504-4.

Matus, F., Rumpel, C., Neculman, R., Panichini, M., Mora, M.L., 2014. Soil carbon storage and stabilisation in andic soils: a review. Catena 120, 102–110. https://doi.org/10.1016/j.catena.2014.04.008.

Mazzarino, M.J., Bertiller, M., Schlichter, T., Gobbi, M., 1998. Nutrient cycling in Patagonian ecosystems. Ecología Austral. 8, 167 – 181. http://hdl.handle.net/20.500.12110/ecologiaaustral_v008_n02_p167.

McDaniel, P.A., Lowe, D.J., Arnalds, O., Ping, C.L., 2012. Andisols. In: Huang, P.M., Li, Y., Sumner, M.E., (eds) Handbook of soil sciences, vol 1, 2nd edn., Properties and processes CRC Press (Taylor & Francis), Boca Raton, FL. 29–48.

Morales, D., Rostagno, M., La Manna, L., 2013. Runoff and erosion from volcanic soils affected by fire: the case of *Austrocedrus chilensis* forests in Patagonia, Argentina. Plant Soil. 370, 367–380. https://doi.org/10.1007/s11104-013-1640-1.

Nanzyo, M., Kanno, H., 2018. Inorganic Constituents in Soil, Basics and Visuals. Springer. 188. ISBN 978-981-13-1214-4. https://doi.org/10.1007/978-981-13-1214-4.

Oades, J.M., 1984. Soil organic matter and structural stability: mechanisms and implications for management. Plant Soil. 76, 319–337. https://doi.org/10.1007/BF02205590.

Oades, J.M., Waters, A.G., 1991. Aggregate hierarchy in soils. Aust. J. Soil Res. 29 (6), 815–828. https://doi.org/10.1071/SR9910815.

Parfitt, R.L., Wilson, A.D., 1985. Estimation of allophane and halloysite in three sequences of volcanic soils, New Zealand. Catena 7, 1–8.

Parfitt, R.L., Saigusa, M., Cowie, J.D., 1984. Allophane and halloysite formation in a volcanic ash bed under different moisture conditions. Soil Sci. 138 (5), 360–364.

Paul, E.A., 2016. The nature and dynamics of soil organic matter: Plant inputs, microbial transformations, and organic matter stabilization. Soil Biol. Biochem. 98, 109–126. https://doi.org/10.1016/j.soilbio.2016.04.001.

Paul, K.I., Polglase, P.J., Nyakuengama, J.G., Khanna, P.K., 2002. Change in soil carbon following afforestation. Forest Ecol. Manag. 168 (1–3), 241–257. https://doi.org/10.1016/S0378-1127(01)00740-X.

Peinado, F.J.M., Navarro, F.B., Jiménez Morales, M.N., Sierra, M., Martínez, F.J., Romero-Freire, A., Rojo, L., Fernández-Ondoño, E., 2016. Long-term effects of pine plantations on soil quality in southern Spain: Effects of pine plantations on soil quality. Land Degrad. Dev. 27, 1709–1720. https://doi.org/10.1002/ldr.2566.

Plaza, C., Zaccone, C., Sawicka, K., Méndez, A., Tarquis, A., Gascó, C., Heuvelink, G., Schuur, E., Maestre, F., 2018. Soil resources and element stocks in drylands to face global issues. Sci Rep-UK 8, 13788. https://doi.org/10.1038/s41598-018-32229-0.

Peng, X., Zhu, Q., Zhang, Z., Hallett, P.D., 2017. Combined turnover of carbon and soil aggregates using rare earth oxides and isotopically labelled carbon as tracers. Soil Biol. Biochem. 109, 81–94. https://doi.org/10.1016/j.soilbio.2017.02.002.

Poeplau, C., Don, A., Six, J., Kaiser, M., Benbi, D., Chenu, C., Cotrufo, M.F., Derrien, D., Gioacchini, P., Grand, S., Gregorich, E., Griepentrog, M., Gunina, A., Haddix, M., Kuzyakov, Y., Kühnel, A., Macdonald, L.M., Soong, J., Trigalet, S., Vermeire, M.L., Rovira, P., van Wesemael, B., Wiesmeier, M., Yeasmin, S., Yevdokimov, I., Nieder, R., 2018. Isolating organic carbon fractions with varying turnover rates in temperate agricultural soils – a comprehensive method comparison. Soil Biol Biochem. 125, 10–26. https://doi.org/10.1016/j.soilbio.2018.06.025.

Poulenard, J., Podwojewski, P., Janeau, J.L., Collinet, J., 2001. Runoff and soil erosion under rainfall simulation of Andisols from the Ecuadorian Páramo: effect of tillage and burning. Catena. 45 (3), 185–207. https://doi.org/10.1016/S0341-8162(01)00148-5.

Raffaele, E., Nuñez, M.A., Eneström, J., Blackhall, M., 2016. Fire as mediator of pine invasion: evidence from Patagonia, Argentina. Biol. Invasions 18, 597–601. https://doi.org/10.1007/s10530-015-1038-5.

Rienzi, E.A., Marchi, A., Rodriguez, G., 2018. Aggregate size, particulate and total organic carbon in different land uses on a sandy loam soil exposed to wind erosion. Adv. Agr. Sci. 6 (3), 95–111.

Rodríguez Rodríguez, A., Arbelo, C., Guerra, J., Mora, J., 2002. Erosión hídrica en Andosoles de las Islas canarias. Edafología. 9 (1), 23–30.

Rodríguez Rodríguez, A., Arbelo, C., Guerra, J., Mora, J., Notario, J., Armas, C., 2006. Organic carbon stocks and soil erodibility in Canary Islands Andosols. Catena. 66, 228–235. https://doi.org/10.1016/j.catena.2006.02.001.

Rostagno, C.M., 1989. Infiltration and sediment production as affected by soil surface conditions in a shrubland of Patagonia, Argentina. J. Range. Manage. 42, 382–385. https://doi.org/10.2307/3899544.

Schiettecatte, W., Gabriels, D., Cornelis, W.M., Hofman, G., 2008. Enrichment of Organic Carbon in Sediment Transport by Interrill and Rill Erosion Processes. Soil Sci. Soc. Am. J. 72 (1), 50–55. https://doi.org/10.2136/sssaj2007.0201.

Six, J., Elliott, E.T., Paustian, K., Doran, J.W., 1998. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. Soil Sci. Soc. Am. J. 62, 1367–1377. https://doi.org/10.2136/sssaj1998.03615995006200050032x.

Six, J., Callewaert, P., Lenders, S., De Gryze, S., Morris, S.J., Gregorich, E.G., Paul, E.A., Paustian, K., 2002a. Measuring and understanding carbon storage in afforested soils by physical fractionation. Soil Sci. Soc. Am. J. 66, 1981–1987. https://doi.org/10.2136/sssaj2002.1981.

Six, J., Conant, R.T., Paul, E.A., Paustian, K., 2002b. Stabilization mechanisms of soil organic matter: Implications for C-saturation of soils. Plant Soil. 241, 155–176. https://doi.org/10.1023/A:1016125726789.

Six, J., Bossuyt, H., Degryze, S., Denef, K., 2004. A history of research on the link between (micro)aggregates, soil biota, and soil organic matter dynamics. Soil Till Res. 79, 7–31. https://doi.org/10.1016/j.still.2004.03.008.

Totsche, K.U., Amelung, W., Gerzabek, M.H., Guggenberger, G., Klumpp, E., Knief, C., Lehndorff, E., Mikutta, R., Peth, S., Prechtel, A., Ray, N., Kögel-Knabner, I., 2018. Microaggregates in soils. J Plant Nutr. Soil Sci. 181, 104–136. https://doi.org/10.1002/jpln.201600451.

Videla, L.S., Rostagno, C.M., Toyos, M.A., 2008. La materia orgánica particulada: comparación de métodos para su determinación y su valor como indicador de calidad de suelos del Chubut. Ciencia del Suelo (Argentina). 26 (2), 219–227.

Yu, Z., Zheng, Y., Zhang, J., Zhang, C., Ma, D., Chen, L., Cai, T., 2020. Importance of soil interparticle forces and organic matter for aggregate stability in a temperate soil and a subtropical soil. Geoderma 362, 114088. https://doi.org/10.1016/j.geoderma.2019.114088.

Zagal, E., Sohi, S.P., Powlson, D.S., 2013. Free and intra-aggregate organic matter as indicators of soil quality change in volcanic soils under contrasting crop rotations. Soil Use Manage. 9 (4), 531–539. https://doi.org/10.1111/sum.12070.

Zhao, F.Z., Fan, X.D., Ren, C.J., Zhang, L., Han, X.H., Yang, G.H., Wang, J., Doughty, R., 2018. Changes of the organic carbon content and stability of soil aggregates affected by soil bacterial community after afforestation. Catena. 171, 622–631. https://doi.org/10.1016/j.catena.2018.08.006.

Zhou, G.Y., Morris, J.D., Yan, J.H., Yu, Z.Y., Peng, S.L., 2002. Hydrological impacts of reafforestation with *eucalyptus* and indigenous species. Forest Ecol. Manag. 167, 209–222. https://doi.org/10.1016/S0378-1127(01)00694-6.