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Erosion proxies in an exotic tree plantation question the appropriate land use in Central Chile

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ABSTRACT

In South-Central Chile, slopes degraded by former erosion have been afforested with exotic tree species since the 1950s for erosion protection. From 1975 on, primary and secondary forests have increasingly been replaced by tree plantations. This practice is often justified by claiming plantations would similarly protect the soil from erosion, even on areas which used to be natural forest. We assessed if plantations offer a comparable level of erosion protection as the natural forests. A six-year-old *Eucalyptus globulus* (Labill.) plantation was compared to an adjacent secondary forest using soil profiles and erosion proxies, i.e. topsoil parameters linked to erosion and ^{137}Cs inventories. These pointed to higher erosion in the plantation: the mean bulk density of the plantation was 22% higher compared to the forest site, the gravel content in the plantation was 61% higher, the organic matter content was 20% lower, the mean thickness of the litter layer was 2.2 cm lower and the total sand content was 20% higher. The soil loss of the plantation was estimated to be between 4.8 cm (mean profile truncation) and 6 cm (^{137}Cs approach). These results clearly hint to the conclusion, that the tree plantations in South-Central Chile might promote soil erosion instead of preventing it. Thus, current land use management practices seem to impose erosion risks on Chilean soils raising concern about the sustainable development within the study site.

1. Introduction

The importance of soil protection becomes apparent when soils lose their beneficial functions due to overexploitation and subsequent soil erosion. From an anthropocentric point of view, erosion threatens the base of human life through declining soil quality and rendering areas unusable for agriculture or forestry (Pimentel et al., 1995). Its prevalence is increasing globally and is associated with increasing material needs of a growing population (Boardman, 2006). Furthermore, deforestation appears to be a driver of erosion, since it accounts for 43% of the eroded area globally (Oldeman, 1992). Of the two billion hectares physically or chemically degraded, 1.1 billion ha are affected by water erosion (Oldeman, 1992). Due to its long-term effects and scale, soil erosion by water is the most important type of physical soil degradation (Pimentel and Kounang, 1998; Wakiyama et al., 2010). Further deterioration is expected in the course of climate change as more frequent extreme precipitation events and fluctuating rainfall intensities may cause disproportionately high water erosion (Nearing et al., 2004; Yang et al., 2003) and off-site effects, e.g. sedimentation or pollution of water bodies (Barra et al., 2005; Litschert and MacDonald, 2009).

At least 18% of South America's land area is affected by soil degradation (Oldeman, 1992). In Chile, the situation today is worse with 48.9% of the land area affected by erosion (CIREN, 2010). It is mainly concentrated between the 4th and the 10th administrative region in Central Chile, i.e. the zone of temperate and Mediterranean forests (Casanova et al., 2013). There, areas eroded from the former land uses, i.e. wheat farming or grazing have been afforested since ~1900 (Armesto et al., 2010; Endlicher, 1988; Otero and Durán, 2006), but in larger scale since the 1950s (Albert, 1906; Oyarzún and Peña, 1995; Toro and Gessel, 1999). Starting in 1974 in the wake of government subsidies of the Decreto Ley 701, exotic tree plantations, e.g. *Pinus radiata* D. Don and *Eucalyptus globulus* Labill., have increasingly replaced native and secondary forest areas (Altamirano et al., 2013; Armesto et al., 2001; Cisternas et al., 1999; Clapp, 1995; Echeverría et al., 2006; Lara et al., 1989; Lara and Veblen, 1993). This replacement was often justified by claiming the erosion protection traits of exotic tree species – even if natural forest was replaced with less diverse plantations in this

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biodiversity hotspot (Braun, 2015; Clapp, 2001; Myers et al., 2000; Toro and Gessel, 1999). Within the Chilean coastal range, native forests have been reduced to less than a tenth of their extent of 1975 (Braun, 2013; Braun and Vogt, 2014). Thus, there are more plantations than natural forests in the 8th region today (CIREN, 2010). Around 80% of these forest losses have been due to forest plantation establishment and were irrespective of slope or climatic conditions (Braun, 2013). The successful stabilisation of historic gully erosion in Central Chile by tree plantations is not without doubt (Iroumé et al., 1994). In Chile's native forests, dense understories protect the topsoils against erosion. In contrast, cutting forests in favour of tree plantations, including the use of heavy machinery during logging and planting, unregulated forest road establishment on private ground, and pesticide use against weeds, compacts the topsoils and leaves them unprotected for years. This provokes erosion, especially during the rainy austral winter period. In compacted and bare top soils infiltration capacity is reduced (Malmer and Grip, 1990), thus increasing runoff generation. Another aspect of machinery use is, that more dispersive subsoil material may be intermixed with the topsoil, which increases the sediment transport off site (Burt et al., 1983). Finally, harvesting and re-planting usually occur after as a little as 8–12 years in Central Chile, leaving little time for the development of an understory, litter layer or replenishment of carbon stocks. However, the plantation establishment is still ongoing: *Eucalyptus globulus* is now the second most planted tree species in Chile and has been increasingly planted since 1990 (Casanova et al., 2013), resulting in the second largest cluster of exotic species in South America (Jobbágy et al., 2012). Forestry is, therefore, a major contributor to Chile's economy (Salas et al., 2016). Tree plantations in the coastal range feature not only evidence of historic erosion, e.g. gullies, but also show recent erosion (Supplementary material 1), e.g. lack of litter layers, crusting, the formation of armour layers, i.e. grus layers, rills and coarse sand accumulations (Endlicher, 1988).

Hence, it appears that the original idea of protecting historically eroded slopes by afforestation has turned into its opposite, i.e. plantations might even promote erosion (Zhou et al., 2002). Hence, the alleged positive effects of the plantations need to be questioned. To easily and quickly assess erosion quantitatively, one can compare erosion proxies such as increased sand contents or decreased organic carbon contents to a reference land use. Complementarily, erosion is estimated as run-off by sediment traps, profile truncation or ^{137}Cs inventories (Loughran, 1989). The ^{137}Cs method was already successfully applied in Chile (Schuller et al., 1997). Within small areas, the ^{137}Cs fallout after global nuclear weapon tests is assumed evenly distributed (Schuller et al., 2003; Walling and Quine, 1991). Erosion can be estimated, given there is a close reference site, where erosion and sedimentation have been as low as possible and the migration of ^{137}Cs has been identical over time (Zapata, 2003). Surplus ^{137}Cs inventories compared to the reference site are explained by sedimentation, whereas inventories below the reference value are related to erosion.

Herein, we compared a secondary forest with a *Eucalyptus* plantation regarding their soil physical parameters as erosion proxies. We estimated total erosion by two methods: profile truncation and ^{137}Cs inventories. The null hypotheses were that the erosion proxies and the total erosion would be nearly identical between the land uses. To the best of our knowledge, this is the first comparison of erosion proxies of natural forest and a *Eucalyptus* plantation in Chile.

2. Material and methods

The study site was in the transition zone of the coastal range and the dry central plain of the 8th administrative region of Chile (Fig. 1), in the community of Ninhue (36.373° S, 72.382° W). The size of the study area was 100 m by 180 m and the average elevation was 140 m above the sea level. The slope was 12.5° (south-facing) and was identical among the land uses. The area drained into the Río Lonquén, which in turn drains into Río Ñuble. It consisted of a secondary forest of *Cryptocarya alba* (Molina) Looser, *Lithraea caustica* (Molina) Hook & Arn., *Aextoxicon punctatum* (Ruiz & Pav.), *Maytenus boaria* (Molina), *Colliguaja odorifera* (Molina) and *Baccharis concava* (Pers.). The forest was surrounded by a 6-year old *Eucalyptus globulus* plantation in the second rotation. The harvest residuals from clear-cutting were aligned along the drop lines. The Köppen-Geiger climate type was a warm-temperate Csb climate with an annual precipitation of 655 l m^{-2}, 80% of the precipitation concentrated between May and August and a mean annual temperature of 14.7 °C (Ramirez, 2002). The parent material of this part of the coastal range was granitic (Casanova et al., 2013; SERNAGEOMIN, 2003). The predominant soils of the range were rather shallow reddish, yellowish cambisols (FAO, 2006; Oyarzún, 1995). In the case of severe erosion, leptosols are abundant, and if dry, these soils are prone to hard setting. In the 8th administrative region, 31.9% of the area were affected by erosion, while in the community Ninhue 51.7% were affected, making it to one of the most erosion-affected communities in the 8th administrative region of Chile (CIREN, 2010). Between 1988 and 2000 alone, the tree plantation area in this community grew from 2385 to 6356 ha, i.e. by 166% (Ramirez, 2002). The sampling took place in September and October 2012.

2.1. Land use

The VII. and VIII. region of Chile are dominated by a transition from native (sclerophyllous and deciduous) forests and shrublands to commercial tree plantations of *Eucalyptus* and *Pinus* (Braun, 2013; Echeverría et al., 2006; Heilmayr et al., 2016). This is also the case for the study site at Ninhue. Until the 1990s, the site featured a mixture of native forests in the moister parts (within sidehill cuts along creeks) and sclerophyllous shrublands along crests and hill flanks. In the 1990s, the latter were cut to establish commercial tree plantations. The site was planted with *Eucalyptus* to the west and *Pinus* to the east. The central parts were cleared for pastures. Forest remained within steep sidehill cuts, where the terrain was not suitable for the machinery required for plantation establishment.

2.2. Soil profiles and erosion proxies

First, one soil profile was comprehensively characterised on each land use (forest, plantation). Comparing these two profiles revealed which soil physical parameters were interesting to compare on a larger scale. To assess the soil physical parameters on a larger scale, 49 samples (21 and 28 in the forest and the plantation, respectively) were taken from random locations: undisturbed topsoil samples were taken from 0 to 5 cm depth using steel sampling cores (5 cm × 2.5 cm radius, 100 cm^3 volume, Eijkelkamp Agrisearch Equipment, Giesbeek, The Netherlands). At each sampling location, the maximum thickness of the litter layer was measured. In smaller pits (~30 cm depth), the depth of the topmost granite residues, i.e. weathering front, was taken as an estimator for the soil profile truncation. Assuming identical erosion or sedimentation between the land uses, the mean depth of these residues should be equal. Shorter distances between the soil surface and the granite residuals indicate profile truncation, i.e. soil loss (Jankauskas and Fullen, 2002).

After drying the samples for 48 h at 60 °C, the bulk density was determined as the dry weight per volume minus the gravel content. Gravel contents were determined as the residuals after dry sieving (2 mm; Gilson Company, Lewis Center, OH, U.S.A.) of the dried and manually disaggregated soil. Sand fractions were collected during wet sieving after destroying the organic matter (OM) by hydrogen peroxide (15%), sodium pyrophosphate solution (15%) and heating up to 45 °C for 1 h. Grain sizes are given in the Krumbein Φ scale (coarse sand 2–0.5 mm; medium sand 0.5–0.25 mm; fine sand 0.25–0.063 mm, Krumbein and Sloss, 1963). The weight loss of 5 g dried and 2 mm sieved soil at 550 °C after 4 h gave the total OM contents. For a subset of 11 samples (forest: n = 6; plantation: n = 5), the aggregate stability

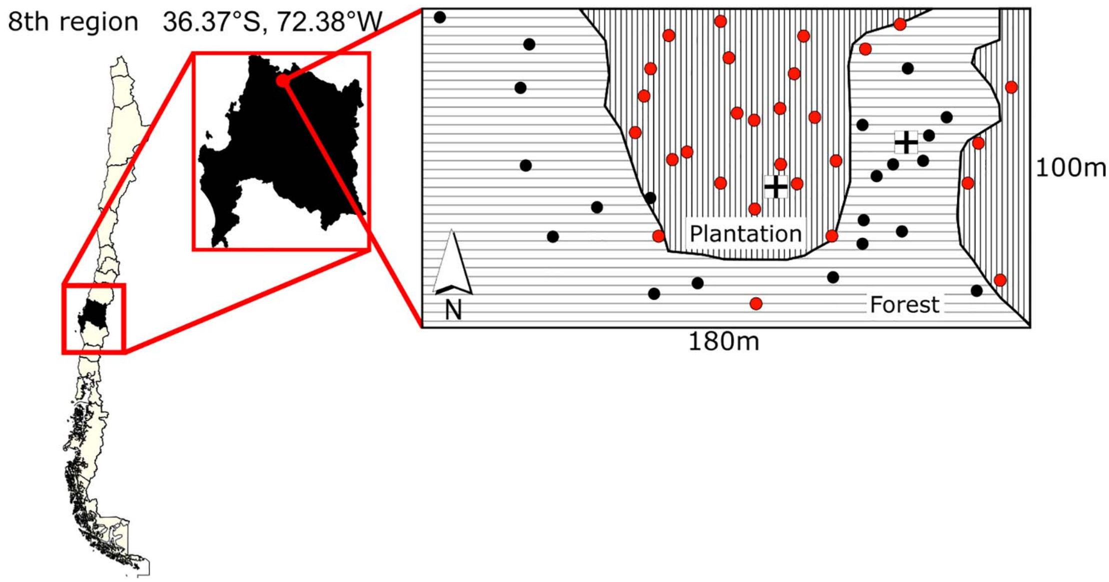

Fig. 1. Schematic representation of the study area. Vertically striped areas represent the plantation, while the horizontal stripes represent the forest site. The line between the land uses is the approximate land use boundary taken from an aerial photo. Red and black dots represent sampling points of the plantation ($n = 29$) and the forest site ($n = 21$), respectively. Pluses are the two main profile locations. Map based on previous work by Giovanni Fasano, distributed under a CC-BY-SA 3.0 license. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

was tested according to Herrick et al. (2001).

The ^{137}Cs bulk samples were taken at three sites: at each of the two soil profiles and at a reference site located uphill on the water divide. At the reference site, no sedimentation was expected because of its location on the water divide and negligible erosion was expected because of the vegetation cover. No signs of erosion were visible. Bulk samples ($20 \times 15 \times 3$ cm) were taken down to 50 cm depth. They were dried at 60 °C for 48 h, ground and put in Petri dishes. The ^{137}Cs activity was measured by gamma-ray spectrometry after calibration against standards (QCYB410 by Eckert & Ziegler Nuclitec GmbH, Germany, for the 122–1836 keV range), using an ORTEC high-resolution, extended range Ge detector of 53% relative efficiency (72,000 s count time per sample giving results with an analytical precision of 10% at the 95% level of confidence). Results were expressed as ^{137}Cs activity per mass (Bq kg^{-1}) in each 3-cm depth increment. These were also normalised to activities per area (Bq m^{-2}) by multiplying the activity per mass with the thickness and the measured bulk density. Summing these activities per area for all increments gave the total areal activity.

2.3. Statistics

All data was grouped according to the land use, i.e. 21 data points for the forest and 28 for the plantation. Outliers were removed after Nalimov's test. Given are means ± standard errors, except for ^{137}Cs data, for which standard deviations were calculated. For all tests, a level of significance of $\alpha = 0.05$ was chosen. For each group, each soil physical parameter was tested for normal distribution using Lilliefors test. In the case of non-normality, Wilcoxon's rank sum test tested for differences between the groups. In the case of normality and equal variances, the two sample *t*-test was used to check for differences in means. In the case of unequal variances, Welch's test was chosen instead.

For clear and easy-to-grasp visualisation of the differences between the two land uses, a kriging interpolation was performed in ArcGIS 10 (Spatial Analyst Toolbox, ESRI, Redlands, CA, U.S.A.). The data were fitted to the ordinary spherical semivariogram model. Automatically calculated parameters were taken for sill, nugget, and range.

3. Results

3.1. Comparison of erosion proxies of the two main soil profiles

In each land use at a representative location (Fig. 1), a pit was dug and the soil profile was characterised. Both main profiles were compared to each other to assess the differences between them. The fully developed litter layer in the forest was 6 cm thick and consisted of mostly sclerophyllous leaves and leaf residues in various decomposition stages (Fig. 2, Table 1), while its plantation's very sparse counterpart was 3 cm thick and consisted of tiny, hardly decomposed branches and dry grass patches. Almost no fine roots were visible in the plantation topsoil, while there was a root mat in the forest. Consequently, the OM content of the topsoil (0–5 cm depth) in the forest was 2.2 times higher than in the plantation (Table 1). The bulk density in the forest was 30% lower and remained rather constant throughout the profile. Aggregates were 25% more stable, while the plantation's profile had 8 times higher gravel and 2.5 times higher coarse sand contents (26%). The total sand content of the forest was 13% higher and fine sand was relatively accumulated. The plantation had a lower total sand content, but the coarse sand was accumulated. In general, the forest's profile B horizon was well structured, very homogenous in colour (5 YR 2.5, profundihumic), high in OM contents and sand, while the plantation's cambic B horizons featured rather differently coloured weathering residues, including quartz grains. The location of topmost granite residuals was in 16 cm depth, while none were identified in the forest site's profile down to − 115 cm. Thus, the A horizon of the plantation appears eroded.

3.2. Soil physical parameters between the treatments

The profound differences between the two main soil profiles (assumed representative for each land use) were statistically confirmed on the entire study area following random sampling (Figs. 3 and 4). On average in the plantation, the OM content was 20% lower and the gravel content was 61% higher than in the forest (Fig. 3; for full quantitative results see Supplementary material 4). The total sand content was 20% higher, the coarse sand content was 28% higher, the medium sand content was 31% higher and also 16% higher fine sand contents were found in the plantation. The bulk density was 22% higher (Fig. 4), the mean thickness of the litter layer was 2.2 cm lower. All

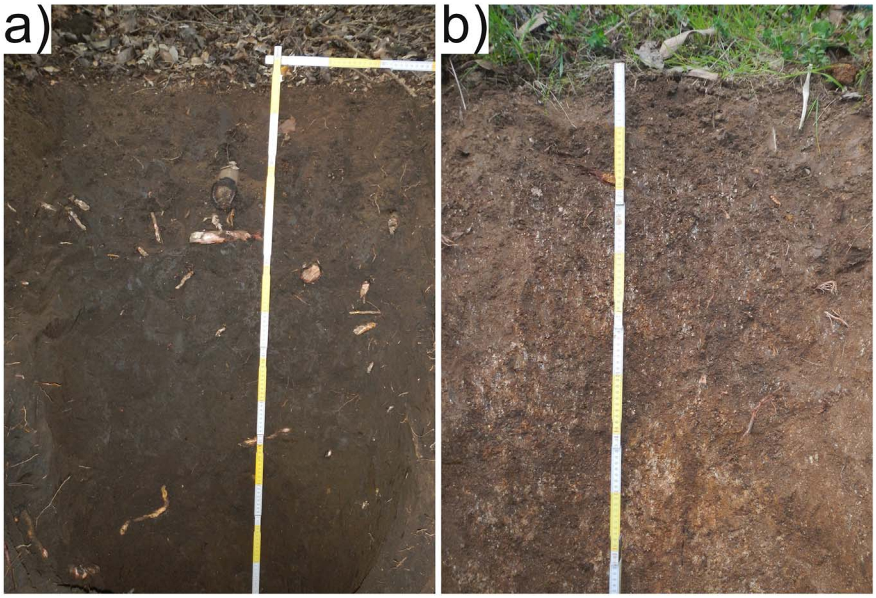

Fig. 2. The two main soil profiles: a) 20-year-old secondary forest (0–115 cm depth, Cambisol (Profundihumic, colluvic, WRB 2014 updated 2015): 6 cm litter layer (sclerophyllous), humic topsoil down to − 10 cm, followed by a darker profundihumic cambic subsoil horizon with a slight increase in sand content and a subpolyedric structure, b) an adjacent 6-year-old *E. globulus* plantation in second rotation (0–85 cm depth, Leptic Cambisol): 3 cm sparse litter layer, a humic topsoil down to − 7.5 cm, followed by two cambic subsoil horizons (down to − 60 cm and − 85 cm, respectively) with increasingly coarse material.

these parameters hint to water erosion in the plantation. This is underlined by the location of the diagnostic horizon (topmost granite residues), which was on average 4.8 cm deeper in the forest site than in the plantation.

3.3. Kriging

The kriging interpolations were carried out to visualise the described differences in the soil physical parameters between the two land uses. They generally featured a high spatial correlation with the land use boundaries (Supplementary material 2): e.g. the interpolation of the bulk density (a) and the litter layer (b) showed high correlations with the land use. According to the interpolation of the litter layer, the effect of the vegetation cover appears to be particularly strong. Generally, there were relocation processes discernible on the slope. No material seems to be washed laterally into the other land use, as there are no kriging indicating such flows.

3.4. ^{137}Cs results

The mean differences in soil physical parameters between the two land uses suggest soil erosion in the plantation, and the mean profile truncation of 4.8 cm would confirm this. With the well-established ^{137}Cs technique, this implication was further corroborated.

^{137}Cs areal activity was 391 Bq m^{-2} (± 31) for the reference site on the watershed, 347 Bq m^{-2} (± 25) for the forest site and 147 Bq m^{-2} (± 14) for the plantation site (Fig. 5; Supplementary material 3). Accordingly, the relaxation depths were 0.0232 m, 0.0226 m, and 0.018 m. No ploughing activity was detected, as no homogenisation of topsoil was apparent. All three treatments showed very comparable decreases with depth, with the topmost data point of the reference graph as an exception. However, this can be easily explained by a sample loss due to a rupture of a single sample container during shipping. Assuming that the depth distribution of ^{137}Cs has been equal on all three sites, the detection limit should supposedly be at a similar depth. The reference's and the forest's trend reach the detection limit of 0.3 Bq kg^{-1} in the depth of 15–18 cm, whereas it was reached at 9–12 cm in the plantation. This can be explained by a truncation of the plantation's profile of 6 cm, i.e. erosion (Fig. 5).

4. Discussion

Despite the large plantation area in South-Central Chile, selecting a study site for the comparison was challenging due to accessibility, heterogeneous land use regimes and histories, and highly variable topography. Comparing the two main soil profiles gave a strong first impression of the effects of the land uses (Fig. 2). These two profiles were representative of the smaller soil pits in the respective land uses and hinted to the main differences, i.e. erosion proxies, found herein and already attributed to erosion in South-Central Chile (Endlicher, 1988; Oyarzún, 1995): OM content, bulk density and grain size distribution (Fig. 3). Most soil physical parameters were not only very different between the two main soil profiles but also significantly different between the land uses and pointed, in fact, to higher erosion in the plantation. These compelling differences also resulted in high spatial correlations with the land use boundaries in the Kriging interpolations (Supplementary material 2). The results may have been even more striking if a primary forest was the reference site.

Distinct erosion signs only identified in the plantation of soil degradation underline these findings (Supplementary material 1): gravel accumulations on the soil surface, dry cracks, pedestals, small rills, armour layers and hard setting in dry conditions are considered meaningful signs of strong erosion (Endlicher, 1988; Stroosnijder, 2005; Ypsilantis, 2011).

Table 1
Comparison of soil erosion proxies of the two main soil profiles in each land use, taken from 0 to 5 cm depth.

Site name	Bulk density [g cm^{-3}]	Gravel [%]	OM [%]	Sand content [%]	Coarse sand [%]	Medium sand [%]	Fine sand [%]	Litter layer [cm]	Aggregate stability (max. 6)	Depth of the diagnostic horizon [cm]
Forest's main profile	0.96	2.0	2.9	57.0	10.4	11.6	35.0	6.0	6.0	> 115
Plantation's main profile	1.24	0.2	1.3	50.5	26.1	9.5	14.7	3.0	4.6	16

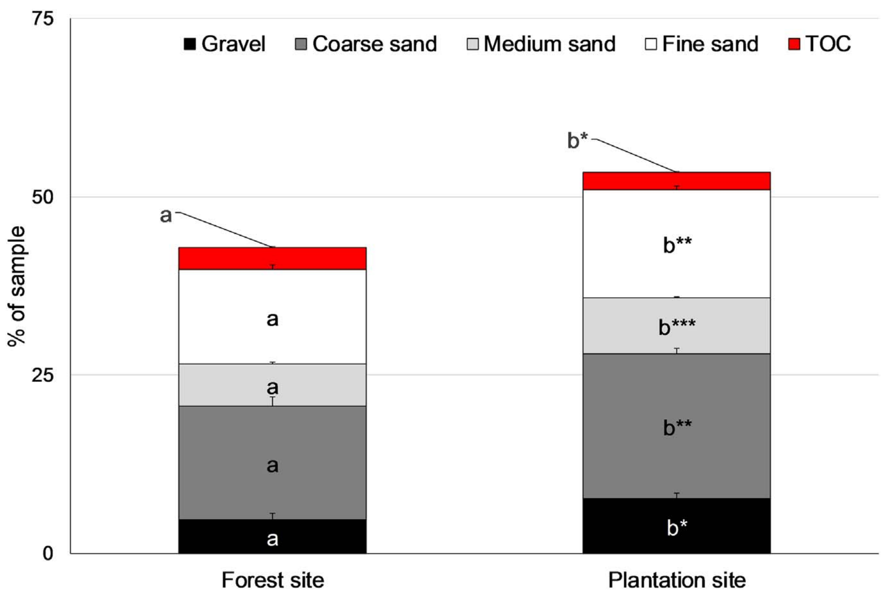

Fig. 3. Contents of the organic matter and the grain sizes: gravel (> 2 mm), coarse sand (2–0.5 mm), medium sand (0.5–0.25 mm), fine sand (0.25–0.063 mm) of the two land uses. Taken from 0 to 5 cm depth, expressed as percent of the sample; means ± standard errors. Different letters for one parameter denote statistical significance. Asterisks denote the level of significance (*α = 0.05, **α = 0.01 and ***α = 0.001).

4.1. *The interactions of water erosion and soil physical parameters in Eucalyptus plantations*

Clear-cutting and re-planting by heavy machinery cause compaction of the roads, skid trails and along the tree rows (Gayoso and Iroumé, 1991). This likely contributed to the significantly higher bulk density in the plantation (Fig. 4), potentially leading to lower infiltration, increased run-off generation, and thus stronger water erosion (Ellies, 1995; Fernández et al., 2004). Especially under the Chilean heavy winter rains, high run-off is expected (Boardman, 2006; Casanova et al., 2013). Armour layers and e.g. significantly higher gravel and coarse sand contents in the plantation (Fig. 3) underline this finding. Increased bulk densities and armour layers were already linked to the management of exotic tree plantations (Ellies, 1995; Gayoso and Iroumé, 1995; Merino and Edeso, 1999).

Run-off causes loss of predominantly clay, silt, OM (Lal, 2003) and also nutrients such as potassium (Dechen et al., 2015). Along with pesticide applications on the anyway bare and disturbed soil during the first years (Focardi et al., 1996; Rubilar et al., 2008) and higher bulk density (Lal et al., 1989; Stirzaker et al., 1996), regrowth of shrubs, grasses and herbs is impeded. This may have two consequences: I) a decrease in litter production, which protects the soil from raindrop impacts (Jobbágy et al., 2012), and, II) starting a positive feedback loop of lower plant C inputs causing decreasing fertility (Paterson, 2003).

Generally, the thicker and denser the litter layer, the more kinetic energy of the raindrops is dissipated (Carneiro et al., 2009; Endlicher, 1988; Zhou et al., 2002). The significantly thinner litter layer in the plantation does not protect the soil as the thicker layer of the forest (Fig. 4). The thickness of the litter is highly correlated with infiltration (Tricker, 1981), but it is a rather weak proxy for the quality of the litter layer and does not fully characterise it, nor its erosion protecting properties. As the maximum thickness was taken in the field, this parameter likely overestimates the litter layer of the plantation. Further studies should, therefore, assess more diverse properties of the litter layer. Contributing to its thickness, *E. globulus* plantations may produce 20% less litter than natural forests (Molinero and Pozo, 2004) because of allelopathic compounds stemming from the trees (Zhang and Fu, 2009) or the high water demand of *E. globulus* (Drake et al., 2012; Forrester et al., 2010). As water scarcity is an increasing challenge in South-Central Chile during the austral summer (Valdés-Pineda et al.,

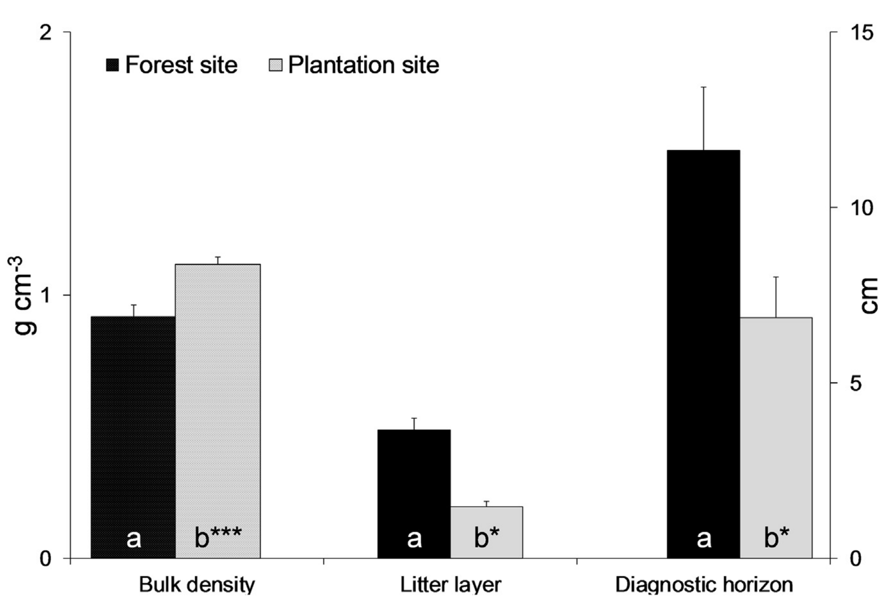

Fig. 4. Bulk density of the topsoil (g cm^3; on left y-axis), thickness of the litter layer and depth of the diagnostic horizon (topmost granite residues; both cm and on right y-axis) of the two land uses, based on 49 data points (21 and 28 for the forest and plantation, respectively). Given are means ± standard errors. Different letters for one parameter denote statistical significance. Asterisks denote the level of significance.

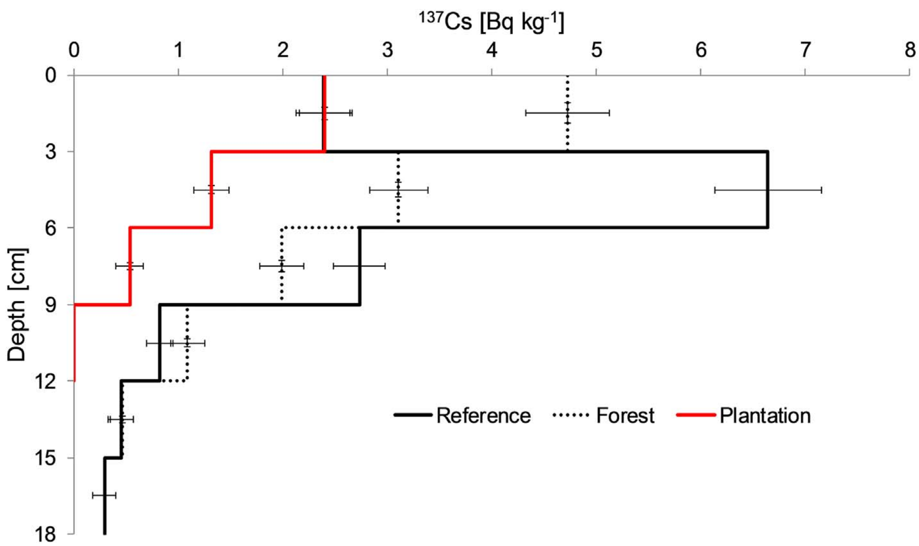

Fig. 5. Depth distribution of the ^{137}Cs activity per mass unit ($Bq\,kg^{-1}$) ± standard deviations in the three locations (reference site on the watershed, secondary forest and *Eucalyptus* plantation).

2014), high water demand of *E. globulus* plantations might enforce unfavourable plant growth conditions in dry years. It is often reported that the ground cover regenerates within two years after plantation establishment, which strongly reduces erosion by up to 90% and increases infiltration (Iroumé and Gayoso, 1989; Oyarzún, 1995). In our case study, however, 6 years after planting no extensive, dense ground cover had developed (Fig. 4, Supplementary material 1). In agreement with this, Braun (2015) and Braun and Vogt (2014) have seldom found dense ground covers among hundreds of tree plantations. So, the growth conditions depend on the plantation management and do not always allow quick re-establishment of ground cover after plantation establishment. Considering 6 years of predominantly unprotected topsoil during rotations of 14–15 years, this raises concerns over the suitability of exotic tree plantations in sloped areas of Chile. Further research should focus on the factors for successful and fast regrowth of the understorey in *Eucalyptus* plantations. Adding to the protection of the soil, aerial photographs clearly hint to a higher canopy density and therefore higher interception in the forest site.

The OM contents were well correlated with the thickness of the litter layer (Pearson's correlation coefficient 0.53). Low C input, dissolved organic carbon loss by erosion and soil microbial respiration lead to decreasing OM contents and nutrient pools over time. OM contents below 2% are linked to increased erosion (Fialho and Zinn, 2012; Oades, 1988). The mean OM content of the plantation is still above this threshold (Fig. 3), but contents might drop after the next clear cutting, thus posing further substantial erosion risk. OM stabilises soil aggregates and increases water infiltration rates (Boix-Fayos et al., 2006; Descheemaeker et al., 2006). This is underlined by the higher mean aggregate stability in the forest site (5.9 out of 6) than in the plantation (5.6 out of 6; Supplementary material 4). Albeit only a subset was tested for aggregate stability, it underlines that the plantation topsoil may be more vulnerable to water erosion. Connected to the turnover of soil organic matter, *Eucalyptus* species may also decrease soil microbial biomass, metabolic activity, and functional diversity, probably by a reduction in understory cover (Behera and Sahani, 2003; Chen et al., 2013; Zhang et al., 2017), therefore negatively affecting the soil microbial functioning, e.g. nutrient cycling.

Consequently, the plantation topsoil is less likely to withstand the erosive power of the run-off. During harvest and replanting more dispersive, low OM content subsoil is mixed with topsoil, which increases the overall erodibility. As water erosion primarily detaches finer grain sizes, the coarse fractions enrich over time: the total sand content and the individual sand fractions were significantly higher in the plantation (Fig. 3). Similarly, a sand accumulation in a *Eucalyptus* rotation was found for a granitic Ultisol in China (Xu et al., 2000). Only if the fine sand fraction was further divided, the very fine sand (0.125–0.063 mm) featured no difference between the treatments (data not shown). In the case of severe erosion, only gravel remains, which was the case in the plantation and eventually the soil profile will be truncated. In the plantation, the gravel contents were most strongly correlated with the profile truncation (correlation coefficient 0.43).

4.2. Comparisons of soil loss

These results are in good agreement with the soil loss estimations: soil loss in the plantation was estimated by two methods, profile truncation (Fig. 4) and ^{137}Cs inventories (Fig. 5). Both methods gave very comparable results and indicated a mean soil loss of about 5–6 cm in the plantation. The ^{137}Cs data needs careful interpretation, as there are open questions: the spatial variability during deposition of ^{137}Cs remains unclear, as it might have been patchy in forested areas. Also, the land use history of the area is not fully known prior to the year 1975. Strong ^{137}Cs losses due to erosion under a different, unknown land use until 1975 might influence the results. However, ^{137}Cs areal activities are plausible and comparable to other studies in the coastal range of Chile, where values in the range of 450–610 $Bq\,m^{-2}$ were reported (Schuller et al., 2002, 2003).

4.3. Implications for sustainable development

Even though for highest erosion protection and biodiversity, native forests are best not converted into exotic tree plantations (Oliveira et al., 2013), this conversion is still ongoing in Chile (Miranda et al., 2017). From an erosion protection point of view, more sustainable plantation management should aim at preventing a negative feedback cycle of OM losses and increased erosion. OM is crucial for erosion protection (Rajan et al., 2010) due to its negative effects on erosion (e.g. enhancing aggregate stability and infiltration capacity) and its positive feedback on soil fertility (e.g. fuelling microbial turnover). As frequent clear-cuts and erosion of bare soil during the austral winters diminish the OM stocks, longer rotations and the promotion of understorey vegetation could increase topsoil OM stocks over time (Jiao et al., 2011). Harvest residues are preferably arranged along the contour lines and not removed. In this regard, weed control during plantation establishment might be detrimental eventually: C inputs and, therefore, OM build-up likely decrease due to lower biomass production of the understorey. Fire clearance after harvest also needs to be considered as an additional detrimental factor of OM levels (Clapp, 1995, 2001). Consequently, a less developed litter layer and more sparsely distributed understorey less likely dissipate the rainfall's kinetic energy. Management should intend to physically disturb the soil as little as possible. A transition from clearcutting towards single tree extraction may be considered optimal. Also, the choice of machinery needs to address soil compaction, as well as mixing of more dispersive, low OM subsoil with topsoil during planting. As a minimum, OM levels and the soil texture are regularly monitored in plantations. Once *Eucalyptus* is established after 2–3 years, its high water demand during the dry austral summers as well as allelopathy might only leave little opportunity for understorey to develop. These properties raise the question of long-term sustainability of *Eucalyptus* plantations for Central Chile.

5. Conclusion

In this case study, we compared erosion proxies of a *Eucalyptus* plantation to an adjacent secondary forest. Since erosion is a complex phenomenon, erosion proxies have to be interpreted cautiously and should be extended in further research. However, applying several erosion proxies which, I) independently lead to a similar interpretation and II) do not indicate only slight, but considerable recent erosion, we are convinced that the results sufficiently provide evidence for stronger erosion in the *Eucalyptus* plantation. So, the current plantation management does not seem to prevent erosion. Instead, the replacement of forests by *Eucalyptus* plantations may, in fact, promote erosion. If further research confirms our findings, an increase in soil erosion by plantations raises concerns regarding the regional soil conservation given the vast areas where this replacement process was observed.

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Conflicts of interest

The authors declare no conflicts of interest.

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