**ESTIMATING SOIL REDISTRIBUTION PATTERNS WITH ^{137}Cs
MEASUREMENTS IN A MEDITERRANEAN MOUNTAIN
CATCHMENT AFFECTED BY LAND ABANDONMENT**

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ABSTRACT

In Mediterranean mountainous environments, the removal of natural vegetation for developing agriculture increased the surface areas prone to erosion in the past centuries. In Southern Pre-Pyrenees, the process was inverted during the middle of the XXth century. This work aims to assess how land use changes after widespread land abandonment affect soil redistribution. For this purpose, ¹³⁷Cs was used in a 23 km² catchment that was mostly cultivated at the beginning of the past century. After land abandonment, 16.5% of croplands persisted but afforestation and natural revegetation occupy 83.5% of the catchment area. ¹³⁷Cs massic activity and related soil properties - stoniness, grain size and organic matter contents- were analysed in 98 bulk core samples. Physiographic characteristics -slope, altitude and solar radiation- at the sampling points were determined by using Geographic Information Systems (GIS). Soil erosion and deposition rates were derived from ¹³⁷Cs measurements after applying conversion models and were spatially interpolated to estimate the amount of net soil loss. In cropland soils mean erosion (62.6 Mg ha⁻¹year⁻¹) and deposition rates (55.2 Mg ha⁻¹year⁻¹) were significantly higher than in the other land uses. The lowest mean erosion rates (2.4 Mg ha⁻¹year⁻¹) were found in natural forests and the lowest mean deposition (2.6 Mg ha⁻¹year⁻¹) in pine afforestation evidencing the soil stabilization achieved in the last decades due to revegetation. A sediment budget with the interpolated rates, result in a specific sediment yield of 4.15 Mg ha⁻¹year⁻¹. These results outline the impact of land use changes on soil redistribution in fragile mountain agroecosystems.

KEYWORDS: Land use changes, catchment scale, ¹³⁷Cs measurements, erosion and deposition rates, soil and physiographic properties.

INTRODUCTION

Over the past centuries human activities have been the main drivers of transformations in ecosystems by converting natural landscapes into agriculture lands (Chauchard *et al.*, 2007; Ellis *et al.*, 2010, 2013). The steep Mediterranean hillslopes provide a good example of such gradual transformation into terraced arable lands. Agricultural deforestation is a local and global environmental issue (Foley *et al.*, 2005). These problems are exacerbated in fragile environments, such as the Mediterranean mountainous regions. These types of sensitive agroecosystems are prone to erosion due not only to climatic conditions but also because of the strong anthropogenic pressure exerted in these areas during the past centuries and their conversion to agriculture land (Romanyà & Rovira, 2011; Bruun *et al.*, 2015; Colazo & Buschiazso, 2015).

The anthropogenic pressure on the soil system through farming and grazing has existed for several centuries in the central mountains of the Spanish Pyrenees. As a consequence, the natural forest was progressively cleared and replaced by croplands (Alonso-Sarría *et al.*, 2016). The problems that these practices have created are particularly striking in mountainous areas and in regions where agricultural land is found accompanying adverse environmental conditions, such as high elevations, steep slopes, shallow soils and dry climatic conditions with heavy rainfall events (MacDonald *et al.*, 2000). Conventional tillage practices disturb and erode the soil surface and expose the less fertile subsoil, which affects physical, chemical and biological soil properties. These changes can then induce loss of soil nutrients which may, over the long term, decrease the soil productivity. Furthermore, the Mediterranean climate is characterized by irregular spatial and temporal distribution of rainfall events along the year of short duration with the occurrence of very intense rainfall followed by long dry

periods (Mariani & Parisi, 2014). These high intensity rainfalls have been identified as one of the main drivers causing soil loss (Martínez- *et al.*, 2002) and increasing the erosion rates relative to tillage erosion (Quijano *et al.* 2016b). Higher frequency of droughts and extreme storm events projected under climate change will likely produce a critical scenario in these fragile environments. Palazón *et al.* (2016) simulated a scenario with a temperature increase of 2 °C that showed different responses in specific sediment yields for the type of vegetation covers with increases for scrubland in a South Pyrenean catchment.

In the past century the industrialization of Europe and the following politics (CAP) applied, triggered socio-economic changes such as a rural exodus, the decline of traditional small-scale agriculture and pastoralism. Since 1960s, these changes were characterised by an abandonment of marginal areas, generally situated in the mountains, where traditional land uses became progressively economically nonviable (Debussche *et al.*, 1999) that leading to substantial landscape changes in the Pre-Pyrenean mountain areas (Lasanta *et al.*, 2016; Quijano *et al.*, 2016c; Navas *et al.*, 2017). Other important indirect impacts of land abandonment are the rapid siltation of water bodies at the first stages of the abandonment before the natural revegetation growth, which decreased reservoir storage capacities. Previous investigations dealing with this issue, Valero-Garcés *et al.* (1997) and Navas *et al.* (2009) identified increases in sediment yields and changes in erosion rates in sediment records accumulated during the last decades in the Pyrenean Barasona and Yesa reservoirs, in NE Spain. The subsequent reforestation that occurred in the 1970s and the 1980s has caused a strong reduction of runoff and sediment connectivity due to vegetation growth that protect soil from erosion (Buendia *et al.*, 2016; Lizaga *et al.*, 2016).

Studies using the ^{137}Cs technique have confirmed its potential over the medium term (~50 years) for estimating soil redistribution rates (Navas & Walling, 1992; Sadiki *et al.*, 2007; Mabit *et al.*, 2008; Alewell *et al.*, 2014) and for validating spatially distributed catchment erosion and sediment yield models (Collins *et al.*, 2001; Mabit *et al.*, 2002; Du & Walling, 2011; Mesrar *et al.*, 2017).

Erosion rates need to be quantified to assess how relatively recent land use changes impacted on soil loss in Mediterranean agroecosystems. To date, there are only few studies using ^{137}Cs to quantify soil redistribution rates at catchment scale (Porto *et al.*, 2003; Mabit & Bernard, 2007; Navas *et al.*, 2013). Furthermore, fewer studies investigated the impact of recent land use changes on soil erosion (Evrard *et al.*, 2010; Gaspar & Navas, 2013; Gharibreza *et al.*, 2013; Navas *et al.*, 2014). Accordingly, we quantify soil redistribution rates using fallout ^{137}Cs measurements and GIS to investigate changes in spatial variability of soil redistribution patterns resulting from land use change associated with vegetation cover in the recent decades. This research would contribute to a better understanding of soil redistribution dynamics in agricultural mountain landscapes and its effect on some of the main soil properties.

MATERIALS AND METHODS

The study catchment

The Barués study area (23 km²) is an ephemeral stream catchment included in the Arba catchment located in the central part of the Ebro Basin (NE Spain) (Figure 1). Rock outcrops in the catchment include two conformable Oligo - Miocene lithostratigraphic units of the Uncastillo Formation dominated by sandstones (Tirapu & Arenas, 1996). The geomorphological setting is clearly conditioned by the low angle dip of the bedding

setting up the path of the streams following the strike of the beds. The climate is Continental-Mediterranean characterized by cold winters and hot and dry summers. Rainfall events mainly occur in spring (April and May) and autumn (September and October) and summer droughts occur between these two humid periods. The area is subject to very intense, though sometimes localized storms. The mean annual temperature is 13.4 °C and the mean annual rainfall is about 500 mm (recorded since 1929 in Yesa reservoir; AEMET). The most abundant soils in the catchment were classified by Machín (EEAD-CSIC) from field surveys (personal communication) as Calcisols and Cambisols (FAO, 2014). The soils developed on Quaternary deposits mainly composed by alluvial deposits have basic pH, low soil organic carbon contents (1.5%) and secondary accumulation of carbonates.

The largest part of the Barué's catchment was cultivated at the down of the 20th century. In 1960s 58% of the area was farmland. Of the total 13.5 km² that was cultivated, as much as 10.2 km² (75%) of the agricultural land had been abandoned for more than 50 years at the time of sampling. At present only 16.5% of the catchment is still cultivated whereas afforestation and natural revegetation areas occupy the remaining 83.5 % (Lizaga *et al.*, 2016). The main land uses are agricultural (16%), pine afforestation (19%), natural forest and scrubland (50%). The main crops are winter cereals (*Triticum aestivum* L. and *Hordeum vulgare* L.). The pine afforestation is mainly composed of *Pinus halepensis* Mill. with a mean tree cover density of 70% (TCD, Copernicus Land Monitoring Services). The natural forest and the scrubland are typical Mediterranean and in many areas are intermixed. They are mainly composed by *Quercus ilex* L., *Quercus coccifera* L., *Rosmarinus officinalis* L., *Genista scorpium* (L.) DC. and

Juniperus communis L.. The scrubland is in the successional stages of natural revegetation, and in the transition for becoming a Mediterranean forest.

Since 1957 a decrease in agricultural land and a transition to natural and revegetated cover have been documented (Figure 1). There has been a decrease in the number of individual cultivated fields but a trend for those fields to increase in size (Figure 2). The change in size and number of fields may be a farmer's attempt to increase the efficiency of production through a decrease in the cost and the quantity of inputs utilized.

Soil sampling

In 2013 a total of 98 bulk core soil samples were collected. A motorized percussion corer equipped with a steel core tube with 40.7 cm² surface area was used to collect two replicates of bulk soil samples at each sampling point from the surface until a depth varying from 20 to 54 cm depending on the local soil thickness.

The sampling points were distributed proportionally across the catchment surface using a previous 500 x 500 m grid created in GIS software to proportionally represent the percentage of surface occupied by the different land uses with a sampling density of 0.23 km²/sample. The grid location was preserved as much as possible although the cores were also taken in representative locations to characterize the condition and properties of the surrounding soil within that land use avoiding recently highly disturbed areas. For each sampling point data on land use, slope angle, altitude and solar radiation were recorded.

Sample analysis and soil redistribution estimates

The two soil cores from each sampling site were mixed in the field and air-dried, grinded, homogenised and sieved to ≤ 2 mm. The fraction > 2 mm was weighed to account for the stone content. Particle size, soil organic matter (SOM) and ^{137}Cs were analysed in the ≤ 2 mm fraction for the 98 soil samples. The Beckman Coulter LS 13320 laser diffraction particle size analyser was used for grain size analysis. Prior to particle size measurements, the samples were mixed with H_2O_2 (10%) and heated (80°C) to remove the organic fraction. Samples were then chemically dispersed with 2 ml of sodium hexametaphosphate (40%), stirred for 2 h and sonication was applied for a few minutes to facilitate dispersion.

SOM was estimated by multiplying SOC content by the Van Bemmelen conversion factor (1.724), assuming that organic matter contains 58 % organic carbon. SOC was analysed by the dry combustion method using a LECO, RC-612 multiphase carbon analyser. The methodology followed for ^{137}Cs analysis is widely described in the literature (Walling & Quine, 1991). The massic activity of ^{137}Cs was measured at the gamma lab of the Experimental Station of Aula-Dei (EEAD-CSIC, Spain) using a high resolution, low energy background, coaxial high-purity germanium (HPGe) gamma-ray detector coupled to an amplifier and multichannel analyser. The detector had an efficiency of 50% and a 1.9 keV resolution at 1.33 MeV (^{60}Co) (shielded to reduce background) and was calibrated using standard soil samples placed in containers of the same geometry as the measured samples. Gamma emission of ^{137}Cs was measured at the 661.6 keV photopeak and counted for 86400s. The analytical precision of the measurements was approximately $\pm 3\text{-}5\%$ at the 95% level of confidence, with a detection limit of 0.3 Bq kg^{-1} . The content of ^{137}Cs was expressed as a concentration or massic activity (Bq kg^{-1}) and as activity per unit area or inventory (Bq m^{-2}). The

inventory was calculated using the mass of the fine fraction and the cross section of the core sampler (Navas *et al.*, 2005).

The values of ^{137}Cs inventory or areal activity density associated with the 98 individual sampling points were converted into estimates of soil redistribution rates ($\text{Mg ha}^{-1} \text{ yr}^{-1}$) by using Soto & Navas (2004) and Soto & Navas (2008) conversion models for uncultivated and cultivated soils, respectively. The models compare the measured inventory with the local reference inventory and determine the erosion or deposition rate required to account for the depletion or increase of the measured inventory, relative to the reference inventory.

For uncultivated soils the model is compartmental and divides the soil into horizontal layers of 1-cm thickness with homogeneous ^{137}Cs distribution in each compartment. A ^{137}Cs linear transference between each pair of successive compartments is supposed, whereby the isotope flux is proportional to the difference in concentration between the two compartments, the proportionality constant being a specific coefficient “k” extracted from reference profiles. In addition to the downward movement, the model also takes into account the ^{137}Cs deposition on the surface. The increases or decreases in concentration level due to ^{137}Cs fluxes are corrected by a factor accounting for the volume of the layers. This volume is taken as the working volume for ^{137}Cs adsorption, which is considered to be the same as that occupied by the soil fraction less than 2 mm.

For ploughed soils, the model has just one compartment that extends from the soil surface to a given cultivation depth (20 – 25 cm in the study area) and assumes a temporal evolution of the ^{137}Cs concentration in the compartment. The ^{137}Cs activity deposited is homogeneously distributed in the compartment within its effective volume.

In order to establish the local reference inventory for the study catchment, two areas were identified as reference sites in flat undisturbed locations under stable soil conditions, where neither erosion nor deposition was expected to have occurred. At these reference sites, 21 soil samples were collected with depths ranging from 25 cm up to 40 cm until the parent material was reached (sandstone strata). Sampling was done using a 40.7 cm² surface area automatic steel core driller and five profiles were sectioned in 5 cm increments in order to study the vertical distribution of ¹³⁷Cs. These soil profiles have been reported in Quijano *et al.*, (2016a). The reference sites were selected to cover both the upper and lower parts of the catchment in order to account for catchment-scale variability in the reference inventory (Navas *et al.*, 2007; Porto *et al.*, 2011).

The statistics of results were analysed using R. Pearson's correlation coefficients analyses were used to assess the relationships between SOM, grain size, ¹³⁷Cs activity and inventory and topographic factors. A Discriminant Function Analysis (DFA) was implemented with R software to identify the variables that better discriminate between the land use groups. A one-way analysis of variance (ANOVA) was performed to assess if erosion and deposition rates were different in function of the land uses. An ordinary kriging with constant trend was selected to model the spatial distribution of the soil redistribution rates at the catchment scale and to spatially represent the erosion and deposition areas (Table 5). The kriging analysis was implemented for the two more homogenous land use distributions separately: the central area (18.2 %) with a predominant agricultural use (95%) and the rest of the catchment (81.8 %) covered by scrubland and forests. To implement the kriging method, we removed eight points in an

area located in the NE part of the catchment which had low variability (Table 5). This approach was justified by the different soil redistribution variances between both areas.

RESULTS

Soil and physiographic characteristics of the catchment

Most agricultural fields are located in the middle part of the catchment at the lowest altitudes. The scrubland and forest had similar slopes and altitudes although the pine afforestation areas occupy the highest altitudes with steep slopes.

Soils were alkaline, non-saline and calcareous. The coarse fraction content varied from 0 to 36% while the SOM contents were low ranging from 0.37 in agricultural land up to 5.93% in scrubland. Most soil samples (64%) had a silt loam texture with proportions of silt varying between 47% and 60%. Twenty-one percent of the soil samples had a loam texture, 14% had a sandy loam texture and the remaining 1% with a 78% of sand had loamy sand texture. More than 75% of the ^{137}Cs massic activity values ranged between 1 and 7 Bq kg⁻¹ and 75% of the ^{137}Cs inventory data varied from 1000 to 3000 Bq m⁻² (Table 1).

Only a small number of the investigated properties were significantly correlated (Table 2). ^{137}Cs was directly correlated with the coarse fraction ($p \leq 0.01$) and most strongly ($p \leq 0.001$) with the organic matter. The SOM was also significantly and directly correlated ($p \leq 0.05$) with the topographic properties, including altitude and slope although negative correlations were found with solar radiation ($p \leq 0.05$).

The mean values of altitude, slope, SOM and ^{137}Cs activity in the agricultural land significantly differed from those under the other land uses ($p \leq 0.01$) (Table 3). In croplands, the ^{137}Cs activity and SOM were lower and significantly different from those

under the other land uses for both erosion and deposition points, with largest differences in deposition points (Table 4). Forest and scrubland had mean altitude and slope significantly different ($p \leq 0.001$) from agricultural that occupied the gentle lowlands and pine afforestation located at higher altitudes with steeper slopes. Although no significant differences were found in the mean clay, silt and sand contents among the different land uses, the agricultural soils were characterised by significantly different ($p \leq 0.05$) smaller means in the coarse fraction at erosion points.

At erosion sites the means of ^{137}Cs massic activity under pine and scrubland were significantly lower ($p \leq 0.05$) than in natural forest. In scrubland and natural forest, the mean of SOM was significantly lower ($p \leq 0.05$) from that in deposition sites of pine afforestation.

The DFA including stoniness, clay, SOM, altitude, slope and solar radiation as selected variables showed better results for erosion (62%) and deposition (83%) samples, when tested separately. In contrast, for the whole sample dataset only 55% of the samples were correctly classified. When ^{137}Cs activity was included in the classification method the percentages of correctly classified samples increased to 61% over the whole dataset, 72% for the eroding sites alone, and 91% for the deposition sites (Figure 3).

Spatial patterns of soil redistribution

The ^{137}Cs reference inventory for the study area was $1507 \pm 92 \text{ Bq m}^{-5}$ estimated based on 21 bulk and sectioned reference profiles (Quijano *et al.*, 2016a). The allowable error was 6.1% at the 95% confidence level (Mabit *et al.*, 2012). As much as 64% of the observations had ^{137}Cs inventories values ranging from 0 to 1505 Bq m^{-2} that were

lower than the local reference inventory. The remaining 36% observations were enriched in ^{137}Cs (range: 1533 to 4230 Bq m⁻²).

The estimated mean soil redistribution rates for Barués catchment were 23.5 Mg ha⁻¹ year⁻¹ and 16.5 Mg ha⁻¹ year⁻¹ for erosional and depositional sites, respectively. Figure 4 represents the magnitude and the spatial variability of the erosion and deposition rates estimated for the individual sampling points over the different land uses. Both erosion and deposition rates were higher in the central axis of the catchment and in the north at the headwaters whereas a lower soil redistribution rate was found in the eastern part of the catchment despite its steeper slopes.

The highest mean erosion and deposition rates were found in agricultural land whereas the lowest were estimated in natural forest. Soil redistribution ranged between 171.4 to 130.45 Mg ha⁻¹ year⁻¹ in erosion and deposition agricultural sites. In forest, erosion and deposition rates ranged between 6.79 to 8.1 Mg ha⁻¹ year⁻¹, respectively.

An analysis of variance showed the occurrence of significant differences between the mean erosion/deposition rates ($p \leq 0.001$) under cropland vs the other land uses (Figure 6). A significant difference ($p \leq 0.05$) was also found between the naturally revegetated forest and the pine afforestation, although no significant differences were found between them and the scrubland in the deposition rates.

The kriging analysis identified that the catchment had up to 76 % of its surface area affected by erosion whereas deposition occurred in the remaining 24 % (Figure 5).

Estimates using the mean soil redistribution values over the isolevel areas extracted from the kriging for the whole catchment amounted to a net soil loss of 9583 Mg year⁻¹ and specific sediment yield of 4.15 Mg ha⁻¹ year⁻¹. The central area with a predominant

agricultural use had a net soil loss more than three times higher than that under scrubland and forest areas (Table 6).

DISCUSSION

Soil and physiographic characteristics in the different land uses

The results have shown significant differences in all soil properties but not in grain size under cropland compared to the other land uses, indicating that this land use is one of the main factors contributing to soil erosion after five decades of land abandonment (Figure 1). The gentler slopes predominating in croplands correspond to a selection of flat terrains to facilitate not only tillage practices, sowing and harvest but also to limit erosion as much as possible. For this reason, the values of altitude, slope, solar radiation, stoniness, SOM and ^{137}Cs activity in the agricultural land differ from the other three land uses.

The absence of significant correlations between ^{137}Cs and the clay fraction are probably due to the limited range of clay contents (80% of the samples had clay content between 5-10%). This result is in agreement with those found in other Pre-Pyrenean catchments by Gaspar & Navas (2013) and by Quijano *et al.* (2016c) in a 1.6 ha cultivated field located in this catchment.

The mobility of ^{137}Cs in soils is primarily controlled by highly selective sorption onto clay mineral surfaces that mostly occurs at the broken edges of the illitic-type clay minerals (Sawhney, 1970; Staunton *et al.*, 2002; Kim *et al.*, 2006). However, SOM is also important in the adsorption of ^{137}Cs , which is supposed to produce a non-specific but a highly efficient mechanism for fixing the radionuclide (Rigol *et al.*, 2002). Moreover, the higher concentration of ^{137}Cs in the fine fraction of stony soils leads to

good correlation between both, stoniness and the radionuclide concentration (Gaspar & Navas, 2013; Zhang *et al.*, 2014). Higher stone volume lower clay volume that results in a higher radionuclide concentration. Likewise, the distribution of the land uses across the catchment may also play a role in the high correlation between ^{137}Cs activity and stoniness. For example, agricultural lands occupy the lowest altitudes and have lower percentages of stoniness, SOM and ^{137}Cs activity. This pattern is generally reported in different environments where zones with higher percentages of vegetation cover are related to high ^{137}Cs activities (Schoorl *et al.*, 2004; Fukuyama *et al.*, 2008; Navas *et al.*, 2011) and lower soil redistribution rates (Navas *et al.*, 2014). In addition to the accumulation of fine particles at the bottom of hillslopes where most cultivated fields are located, tillage practices contribute to the breakdown of soil aggregates. Moreover, the manual clearing of stones to improve the conditions of the cultivated fields also contributes to the reduction of stoniness. Accordingly, higher stoniness is found at higher elevations where the natural and afforested areas dominate.

The higher SOM content in pine afforested and natural forest areas are due to the higher density of the vegetation cover and greater size of the trees resulting in larger aerial biomass. The lower SOM content in scrubland are likely because of the less mature plants. In general, scrub areas were abandoned more recently than natural forest areas, although poorer soil conditions at the time of abandonment cannot be totally excluded. Both ^{137}Cs and SOM were lowest under cropland, which could both be related to the higher erosion rates that lead to the preferential export of fine particles enriched in ^{137}Cs and organic matter (Navas *et al.*, 2014). Our spatial analysis indicated a similar distribution of ^{137}Cs and SOM suggesting both are influenced by similar redistribution

processes. A similar result was found by Navas *et al.* (2011) and Quijano *et al.* (2016a) in similar environments.

The positive correlation of SOM with slope and altitude and the negative correlation with the solar radiation have also been observed by several authors in other mountainous areas (Dorji *et al.*, 2014; Ohtsuka *et al.*, 2008). Altitude and slope correlate with SOM because of the past and present spatial distribution of the cultivated fields. The lands that were first abandoned were located in highest and steepest areas that are the farthest and the less accessible from the village. The early abandonment of these higher areas as cultivated fields and their subsequent revegetation likely contributed to their higher organic matter increment (Navas *et al.*, 2012).

The inverse correlation of SOM with solar radiation is probably explained by the fact that in the less radiated areas the temperature is lower and leads to higher levels of moisture and available water, thus aids the increment of the aerial biomass (Lizaga *et al.*, 2016). The correlation of stoniness with altitude and the inverse correlation of the slope with clay content were dependent on the land use distribution. However, higher erosion rates occurring on steeper slope areas also facilitate the movement of the fine grain particles.

The DFA analysis showed the usefulness of using ^{137}Cs to discriminate between different land uses. The lower discrimination percentage between forest and scrubland is probably explained by the fact that natural forest in many areas is intermixed with scrubland and its characteristics are not so different with woodlands in this intermediate successional status of the vegetation cover. Other tracers as Compound-Specific Stable Isotopes (CSSI) may offer potential to further discriminate among vegetation covers. This technique has been successfully tested in recent research to discriminate land uses

as sediment sources (Blake *et al.*, 2012; Alewell *et al.*, 2016; Upadhayay *et al.*, 2017) in different parts of the world.

Soil redistribution rates

The principal factor affecting the soil redistribution rates is the change of land use that occurred in the last decades. This leads the significant differences in the means of soil redistribution ($p \leq 0.01$) between cropland and the other land uses found in both deposition and erosion areas (Table 3). In agreement with these findings, Navas *et al.* (2013) also found the highest rates of soil redistribution under cropland, whereas the lowest rates were found on forests and scrubland in nearby catchments. The key role of agriculture as the main driver of soil erosion is recognized in a variety of environments (Navas & Walling, 1992; Quine *et al.*, 1994; Mabit *et al.*, 2002; Navas *et al.*, 2014). In addition, the significant higher erosion rates of pine afforestation than in natural forest and scrubland suggest that these natural covers in some areas could better protect the slopes from erosion processes.

Although erosion rates in our catchment are not significant different ($p > 0.05$) between forest and scrubland this is not the case in other more humid environments. In contrast with our findings Navas *et al.* (2005) report in an abandoned Pyrenean catchment ^{137}Cs estimates that discriminate highly eroded scrubland in comparison with the naturally revegetated forest. Navas *et al.* (2008) concluded that this relationship was due to the much poorer soil conditions in the scrubland along with the steepest slopes and the higher solar radiation.

The introduction of additional tracers should be tested in future research to assess the possibility of discriminating differences in soil redistribution rates at different temporal

scales. The $^{210}\text{Pb}_{\text{ex}}$ could offer such potential (Mabit *et al.*, 2014; Porto *et al.*, 2016; Gaspar *et al.*, 2017) in correspondence with the time of the vegetation recovery in the succession status of the revegetation from scrubland to forest in relatively recently abandoned areas.

More mature vegetation is located at the upper parts of the catchment as the result of the early abandonment of the agricultural land. This fact is associated with lower soil erosion and deposition rates in this upper part compared to the central part of the catchment. The introduction of pine afforestation has also reduced soil redistribution rates.

The effect of soil redistribution processes on SOM dynamics is demonstrated by the differences between SOM contents found in the eroded and depositional sites as the higher the erosion rates the lower SOM contents. However, in the natural forest sites this relation is not observed. This is probably due to high stability under forest as identified by the lower redistribution rates that only vary between -2.43 to $3.93 \text{ Mg ha}^{-1} \text{ year}^{-1}$, therefore, soil movement is not as high as to produce a marked contrast in SOM contents.

The results of the sediment budget for the study catchment evidence that higher soil redistribution rates are concentrated in the relatively small central agricultural area and do not occur over most of the catchment. The specific sediment yield in our catchment is comparable with values of $7.56 \text{ Mg ha}^{-1} \text{ year}^{-1}$ obtained in nearby small endorheic catchment (Navas *et al.*, 2014). A similar specific sediment yield of $4.7 \text{ Mg ha}^{-1} \text{ year}^{-1}$ was computed in the Barasona reservoir catchment (Palazón *et al.*, 2016). However, Porto *et al.* (2014) estimated higher mean annual soil loss of $26.39 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ in a small-cultivated catchment (0.86 ha) in Sicily. The kriging approach divided the

catchment in headwater areas mainly composed by woodlands and scrubland and the central part of the catchment composed by croplands, is presented as an alternative for the areal representation of soil redistribution rates over the different land uses. The study area has quite homogeneous lithology and landforms; however, a large variety of land uses may represent a constraint in the use of this methodology such as where the area has a complex land use mosaic. In our case, this approach has some benefits as it allows the differentiation between croplands that concentrate in the lower part of the catchment and scrubland and forests located at the highest altitudes.

CONCLUSIONS

This study demonstrated the potential of ^{137}Cs measurements to quantify and spatialize information on soil redistribution rates in the context of the land use changes in the last decades. The application of the ^{137}Cs method to calculate soil redistribution enabled us to identify and discriminate the main erosion and deposition areas in the catchment under the different land uses. Some of the main factors triggering erosion are related to tillage or farming practices in general, whereas the reduction in erosion rates is related with the increase in the vegetated covers and its maturity.

The results from this study case demonstrate relationships between SOM and soil redistribution rates in cropland, scrubland and pine afforestation areas. The absence of such relationships in forest is likely linked to greater stability and lower soil redistribution rates. The tillage practices and the fact that soil is left bare during part of the year of the predominant agricultural land use in the lowlands led to the higher soil redistribution rates. The lower rates of soil erosion were found under the land uses with more abundant vegetation on higher altitudes and slopes. Therefore, in this

environment, land use was found to be the main controlling factor of soil redistribution rates.

Natural revegetation and reforestation in recent decades after land abandonment produced substantial changes that prevent the erosion and the soil loss. Natural Mediterranean open forests and pine afforestation occupying the highlands, intermixed with scrublands on high to moderate slopes efficiently protect the soil surface from erosion.

Although the ^{137}Cs method generates precise spatial distribution data, the extrapolation at catchment scale is complex but allow to gain a better understanding of the spatial extent, the severity of soil loss over a catchment and the benefits of the natural revegetated areas.

The tentative sediment budget calculated for the Barué's catchment could provide valuable information for implementing soil erosion control programmes in Mediterranean mountains agroecosystems. The present study improves the current knowledge on the relationships between the land use change and the spatial variability of soil redistribution, which may help to mitigate soil degradation, reservoir siltation and implement erosion control practices.

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Table 1

Basic statistics of the different properties found in soils of Barués catchment

	Altitude m	Slope °	Solar radiation WH/m ²	Stoniness %	Clay %	Silt %	Sand %	SOM %	¹³⁷ Cs activity Bq kg ⁻¹	¹³⁷ Cs Inventory Bq m ⁻²
Median	764.68	12.80	6059.90	7.70	8.60	53.40	38.50	2.57	3.70	1314.80
Mean	756.51	14.83	5952.71	9.27	8.93	53.39	37.68	2.50	3.90	1355.10
SD	98.57	9.03	341.17	8.49	2.42	9.69	11.51	1.25	2.40	722.7
Max	932.74	43.37	6393.00	36.45	17.50	81.70	78.00	5.93	9.6	4230.91
Min	554.90	1.27	4724.50	0.00	3.80	18.20	5.90	0.37	0.00	0.00
CV	0.13	0.61	0.06	0.92	0.27	0.18	0.31	0.50	0.61	0.55

Table 2

Pearson correlation coefficients between soil and physiographic properties at the sampling points.

n = 98	Altitude m	Slope °	Solar radiation WH/m ²	Stoniness %	Clay %	Silt %	Sand %	SOM %	¹³⁷ Cs activity Bq kg ⁻¹
Altitude									
Slope	0.0442								
Solar radiation	0.2642	<u>-0.7449</u>							
Stoniness	<u>0.4484</u>	0.1297	0.0694						
Clay	-0.0042	-0.2171	0.1208	<u>-0.3686</u>					
Silt	-0.0462	-0.1	0.028	<u>-0.4539</u>	<u>0.6951</u>				
Sand	0.0396	0.1298	-0.049	<u>0.4598</u>	<u>-0.796</u>	<u>-0.9885</u>			
SOM	0.252	0.241	-0.2095	<u>0.2942</u>	-0.1061	-0.1527	0.1507		
¹³⁷ Cs activity	0.2434	<u>0.2757</u>	-0.2178	<u>0.3248</u>	-0.1157	-0.1356	0.1385	<u>0.6068</u>	
¹³⁷ Cs Inventory	0.0424	0.0798	-0.0768	0.0712	-0.0053	-0.0468	0.0405	<u>0.2881</u>	<u>0.7825</u>

Bold numbers indicate statistical significance at $p \leq 0.05$ level. Bold and underlined numbers indicate statistical significance at $p \leq 0.01$. Bold, underlined and italicized numbers, indicate statistical significance at $p \leq 0.001$.

Table 3

Basic statistics of the soil properties and physiographic characteristics of the sampling points under the different land uses in the study catchment

		n	Altitude m	Slope °	Solar radiation WH/m ²	Stoniness %	Clay %	Silt %	Sand %	SOM %	¹³⁷ Cs activity Bq kg ⁻¹	¹³⁷ Cs Inventory Bq m ⁻²
Agricultural	Median	20	671.84	5.85	6113.40	3.91	8.75	54.85	35.85	1.17	1.98	1427.47
	Mean		690.30	7.51	6129.94	5.32	9.12	53.38	37.51	1.37	2.20	1558.72
	SD		93.31	4.19	84.09	6.17	2.94	9.93	12.12	0.64	1.60	943.66
	Max		847.32	20.24	6333.60	23.03	17.50	65.50	78.00	3.62	7.98	4230.91
	Min		554.90	2.20	6015.20	0.00	3.80	18.20	23.00	0.66	0.00	0.00
	CV		0.14	0.56	0.01	1.16	0.32	0.19	0.32	0.47	0.73	0.61
Forest	Median	15	768.62	20.21	5901.90	9.28	9.20	52.30	39.40	2.98	5.68	1696.19
	Mean		750.65	17.75	5843.61	10.79	9.24	51.87	38.88	3.06	5.86	1718.91
	SD		93.21	7.07	228.11	7.41	2.06	8.42	9.38	0.83	2.03	591.30
	Max		863.36	25.67	6237.60	23.60	14.50	71.60	54.20	4.29	9.50	2970.45
	Min		557.04	2.65	5384.20	2.62	6.20	38.90	17.50	1.55	2.35	821.29
	CV		0.12	0.40	0.04	0.69	0.22	0.16	0.24	0.27	0.35	0.34
Pine	Median	15	826.64	19.43	6066.10	11.06	8.00	47.40	44.10	3.43	5.25	1446.03
	Mean		828.53	20.25	6012.70	12.45	7.93	50.23	41.84	3.15	4.33	1212.15
	SD		61.71	9.49	286.60	9.48	2.17	11.12	13.05	1.28	2.94	799.52
	Max		932.74	35.71	6350.40	35.84	13.30	71.80	65.30	4.90	9.55	2669.63
	Min		706.39	6.82	5419.70	1.65	3.90	30.80	14.90	0.67	0.00	0.00
	CV		0.07	0.47	0.05	0.76	0.27	0.22	0.31	0.41	0.68	0.66
Scrubland	Median	40	757.04	13.98	6049.65	7.86	8.65	53.65	37.90	2.73	3.94	1211.81
	Mean		763.41	15.27	5894.21	9.44	9.07	54.85	36.08	2.60	3.86	1172.96
	SD		96.07	9.19	419.03	8.93	2.36	9.49	11.34	1.26	2.12	592.39
	Max		926.37	43.37	6393.00	36.45	16.60	81.70	58.60	5.93	9.19	2545.57
	Min		583.05	1.27	4724.50	0.00	5.70	34.60	5.90	0.37	0.00	0.00

CV 0.13 0.60 0.07 0.95 0.26 0.17 0.31 0.48 0.55 0.51

Table 4

Mean values of soil properties and physiographic characteristics at erosion/deposition points under the different land uses

EROSION	n	Altitude m	Slope °	Solar radiation WH/m ²	Stoniness %	Clay %	Silt %	Sand %	SOM %	¹³⁷ Cs activity Bq kg ⁻¹	¹³⁷ Cs Inventory Bq m ⁻²
Agricultural	11	689.37	5.38	6110	4.21	8.75	52.46	38.80	1.19	1.42	986.54
Forest	7	734.60	16.39	5781	11.40	9.57	51.07	39.36	3.21	4.35	1223.49
Pine	8	847.13	15.76	5982	13.16	8.10	51.31	40.59	2.74	2.78	717.27
Scrubland	36	759.71	17.06	5828	9.30	9.14	54.81	36.05	2.53	3.00	936.22

DEPOSITION	n	Altitude m	Slope °	Solar radiation WH/m ²	Stoniness %	Clay %	Silt %	Sand %	SOM %	¹³⁷ Cs activity Bq kg ⁻¹	¹³⁷ Cs Inventory Bq m ⁻²
Agricultural	9	691.45	6.53	6154	5.68	9.58	54.50	35.92	1.58	3.15	2258.04
Forest	8	764.70	12.67	5897	10.26	8.95	52.58	38.46	2.92	7.18	2152.40
Pine	6	800.63	12.60	6044	11.38	7.68	48.60	43.72	3.76	6.67	1954.47
Scrubland	12	774.52	15.99	6090	9.85	8.85	54.98	36.17	2.82	6.44	2070.29

Table 5

Main kriging properties for both, agricultural and scrubland and forest areas.

	Range	Nugget	Partial sill	Area (km²)	n
Agricultural	4,994	207	0	4.2	18
Scrubland and forests	8,464	0.5	0.04	18.9	72

Table 6

Ranges of the isolevels curves of soil redistribution rates in erosion and deposition areas used to estimate the net soil loss and gain in the catchment.

		Soil redistribution rates			Area	areal value	
		Maximum	Minimum	Mean	ha	Mg year ⁻¹	Mg ha ⁻¹ year ⁻¹
Agricultural	Erosion areas	170.0	152.9	161.4	1.5	240.0	
		152.9	88.6	120.7	9.8	1182.8	
		88.6	50.5	69.6	16.8	1169.3	
		50.5	27.7	39.1	47.8	1867.7	
		27.7	14.2	21.0	101.5	2131.7	
		14.2	6.2	10.2	67.0	685.4	
		6.2	1.4	3.8	40.8	155.8	
	Total soil loss					7192.7	25.4
	Deposition areas	-0.4	7.6	2.6	40.4	104.3	
		6.6	20.1	13.4	64.6	862.1	
		20.1	42.8	31.5	26.6	837.9	
		42.8	130.8	61.9	6.3	389.7	
	Total soil deposition					2194.0	17.0
Net soil loss						4844.5	
Specific sediment yield							8.3
		Maximum	Minimum	Mean	ha	Mg year ⁻¹	Mg ha ⁻¹ year ⁻¹
Erosion areas		22.9	11.2	17.0	89.1	1516.6	
		11.2	5.8	8.5	240.2	2032.5	

Scrubland and forest	5.8	3.3	4.5	270.1	1224.8	
	3.3	2.2	2.7	232.7	635.8	
	2.2	1.6	1.9	111.6	212.7	
	1.6	1.4	1.5	73.5	112.3	
	1.4	0.9	1.1	134.1	153.9	
	0.9	0.0	0.4	282.4	90.9	
Total soil loss					5979.6	5.2
Deposition areas	0.2	2.7	1.5	312.9	463.0	
	2.7	8.1	5.4	143.8	778.0	
Total soil deposition					1241.0	2.7
Net soil loss					4738.6	
Specific sediment yield						2.5

Figure 1

Location of the study catchment in the central part of the Ebro Basin (NE Spain) (a). (b) Percentage of the different land uses in 1957 and 2016. (c) Pictures of the main land uses in the study catchment. (d) 3D picture of Barués catchment created with a DEM and an orthophoto (National Plan of Aerial Orthophotography, IGN).

Figure 2

Comparison of the variations in size of the cultivated fields (delineated) between 2012 (left picture, IGN orthophoto) and 1957 (right picture, American aerial photograph) of an area located at the central part of the catchment.

Figure 3

Discriminant function analysis for all study points and for erosion and deposition points in the catchment.

Figure 4

Soil redistribution rates at the sampling points of the Barués catchment under the different land uses. Rates are estimated from ^{137}Cs activities (a) and inventories (b) after applying conversion models.

Figure 5

Spatial distribution of the erosion (a) and deposition (b) rates and the corresponding isocurves in the study catchment.

Figure 6

Boxplots of the soil redistribution rates under the different land uses over erosion and deposition areas in Barués catchment.

d

Erosion

Deposition