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The role of silvicultural systems and forest types in preventing soil erosion processes in mountain forests: a methodological approach using Caesium-137 measurements

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24 **The role of silvicultural systems and forest types in preventing soil erosion processes in mountain forests. A**
25 **methodological approach using Caesium-137 measurements**

26

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35 **Abstract**

36 *Purpose* - Forests play a key role in providing protection against soil erosion. Particularly, the role of vertical forest
37 structure in increasing rainfall interception capacity is crucial for mitigating raindrop impact and reducing splash
38 and rill erosion. For this reason, studies on the relationships between forest structures, past management, and the
39 observed rates of soil loss are needed. In the last decades, importance was given to the use of ^{137}Cs as radioactive
40 tracer to estimate soil erosion rates. The ^{137}Cs technique is linked to the global fallout of bomb-derived
41 radiocaesium which occurred during a period extending from the mid-1950s to the late 1970s.

42 *Materials and methods* - ^{137}Cs technique, providing long-term retrospective estimates, could be related to forest
43 treatments applied during the last decades in different sites, also considering the tree species composition. This
44 approach could be useful to compare the effect of different canopy cover and biomass on soil erosion rates related
45 to different tree species. In the work proposed here, a study area dominated by pine and beech high forests located
46 in the Aspromonte mountain (Calabria, Italy) was selected. The measurements, related to forest structural traits,
47 focusing on canopy cover and biomass, and also on management approaches and forest types, are compared with
48 rates of soil erosion provided by ^{137}Cs .

49 *Results and discussion* - The overall results suggest that minimum values of soil loss are documented in areas with
50 higher canopy cover and biomass evidencing the protective effect provided by forests against soil erosion. Also,
51 techniques based on the use of tracers like ^{137}Cs proved to be helpful to select the best forest management options
52 useful to optimize the protective role of forests, with the aim to reduce erosion processes in a long-term perspective.

53 *Conclusions* – However, the experiment indicates that care must be taken when new silviculture treatments are
54 planned. These findings are in agreement with what documented by other authors in similar environments but need
55 further studies to confirm the effectiveness of using ^{137}Cs in different forest ecosystems.

56

57 **Keywords** Apennine forests • Beech forests • Caesium-137 • Calabrian pine forests • Silviculture applied • Soil
58 erosion rates

59

60 **1 Introduction**

61 Forests play a key role in providing protection against runoff and soil erosion (Miura et al. 2003). Soil erosion is
62 a relevant phenomenon in pedogenesis process and affects the evolution of natural ecosystems. Ecological factors
63 like canopy cover, tree species, forest vertical stratification, but also different types of forest management are
64 crucial in increasing rainfall interception and reducing the magnitude of soil loss (Elliot et al. 1999; Hartanto et al.
65 2003). In the last decades, several studies carried out in forest environments stressed the importance of
66 hydrological functions (Bosch and Hewlett 1982; Moore and Wondzell 2005) and their relationships with soil
67 degradation (Pennock and Van Kessel 1997). The most serious risk, however, is related to soil erosion by water,
68 which is one of the most pressing environmental threats worldwide (Reich et al. 2000). Examples in Mediterranean
69 areas are reported, among the others, by Conacher and Sala (1998) and Montanarella (2007).

70 The effort to mitigate soil erosion is a prerequisite to provide the self-preservation of forest areas, in order to
71 perform a sustainable forest management. This topic was widely studied in Italy by several researchers (Zanchi
72 1988; Ollesch and Vacca 2002; Bagarello et al. 2010) and experiments dealing with soil erosion measurements are
73 also available (Alcamo et al. 2007; Sanchis et al. 2008). However, despite the utility of studies carried out in
74 experimental plots and catchments to quantify runoff and soil loss, some difficulties to extrapolate the results in
75 larger areas where no direct measurements are available have emerged. These difficulties are mainly related to the
76 scale effect in the attempts to extrapolate across the landscape modelled rates of water erosion based on plots (see
77 Evans and Boardman 2016; Evans et al., 2016).

78 The need to test alternative approaches to collect more information on larger areas suggested the use of Caesium-
79 137 (^{137}Cs) to provide reliable estimates of soil erosion both at plot (Porto and Walling 2012a,b) and at catchment
80 scale (Porto et al. 2001; 2009). ^{137}Cs is a manmade radionuclide used for soil erosion assessments in many areas
81 of the world (Zapata et al. 2002; Mabit et al. 2008; Golosov et al. 2013; Alewell et al. 2014) and its feasibility for
82 forest areas has been revealed (Garcia-Oliva et al. 1995; Porto et al. 2001; 2009; Estrany et al. 2016). The use of
83 ^{137}Cs offers a great potential to provide retrospective information on medium-term average of soil loss on the basis
84 of a single sampling without the need to disturb the system by installing measuring equipment. The ^{137}Cs technique
85 makes use of the global fallout of bomb-derived radiocaesium which occurred during a period extending from the
86 mid 1950s to the late 1970s. In most environments, the ^{137}Cs fallout reaching the land surface was rapidly adsorbed
87 by the surface soil and its subsequent redistribution within the landscape will have occurred in association with
88 the erosion, transport and deposition of soil and sediment particles. For forest environments, Porto et al. (2001;

89 2014) demonstrated the feasibility of the ^{137}Cs approach by comparing estimates of soil erosion provided by the
90 latter with long-term measurements of sediment yield available in small catchments afforested with eucalypt
91 (*Eucalyptus occidentalis* E.). However, to date, no information on the relationships between ^{137}Cs and parameters
92 like canopy cover, tree species, forest vertical stratification and types of forest management are reported.

93 In this work, a set of four experimental plots in beech and Calabrian pine forest stands was established with the
94 aim to study the relationships between different types of forest management and the observed rates of soil loss
95 using ^{137}Cs .

96

97 **2 Materials and methods**

98 2.1 Study area

99 The study area is located in Ferraina, within the Aspromonte National Park, Calabria, Southern Italy (**Fig. 1**). In
100 this site, four rectangular plots (15 m x 20 m), with slope values ranging from 9% to 22%, were established in
101 2016 (**Fig. 2**). The choice related to the rectangular shape of each plot (with the longer axis parallel to the flow
102 direction) was made with the aim to compare the results of this experiment with those that can be provided by
103 other models like RUSLE (and derived versions) that explore the same range of length and slope.

104 The forest cover consists of two Calabrian pine (*Pinus nigra laricio* Poir.) stands, with different canopy cover and
105 living biomass amounts, and two beech (*Fagus sylvatica* L.) stands (one of which was cut in 2009). More
106 particularly, the first two plots, namely 'Pine stand 1' and 'Pine stand 2', are covered by Pine forests (ca. 50 years
107 old) planted in the late 1960s during a massive re-afforestation programme (ca. 150,000 ha) carried out in Calabria.
108 The remaining two plots, namely 'Beech stand 3' and 'Beech stand 4', consist of semi-natural high Beech forests
109 (ca. 140 years old) typical of the mountain areas of the Mediterranean basin. Beech stand 4 was substantially
110 undisturbed by cutting activities, with the exception of minimal removal of trees in the past decades; on the
111 contrary, Beech stand 3 was clearcut in 2009 and then left to evolve naturally.

112 A reference site with no tree coverage was also selected within an undisturbed flat area, located nearby the forested
113 plots, in order to collect information on ^{137}Cs fallout on the overall study area. The elevation of the study area
114 ranges from 1366 to 1465 m a.s.l.. The mean value of annual rainfall, based on the data obtained at Gambarie
115 d'Aspromonte meteorological station (38°07'36''N, 15°57'08''E - 1300 m a.s.l.) and available for the period 1931-

116 2016, is 1611 mm, with a mean value of annual temperature of 10.6 °C. The climate of the area is classified as
117 oceanic temperate. The major soil types are characterised by a silt-loam texture (Soil Survey Staff 1993).

118

119 2.2 Soil sample collection for ¹³⁷Cs analysis

120 Soil samples were collected in 2016. A detailed sampling programme was established to provide long term data
121 useful to estimate erosion and deposition rates according to the amount of ¹³⁷Cs in the soil. A total of nine replicate
122 soil cores was collected for each plot, using a 10-cm-diameter steel core tube inserted to a depth of ca. 42 cm.
123 More particularly, three transects aligned with the line of steepest slope were established for each plot. In the
124 central transect, three replicate sectioned cores (each one consisting of 14 x 2 = 28 layers) were collected in order
125 to obtain information on cesium distribution in soil. Six additional replicate bulk cores were also collected from
126 the remaining two transects (see **Fig. 2**).

127 Even if several studies have emphasized the effect of forest canopy on fallout interception (see among the others
128 Bunzl et al. 1995; Strebl et al. 1999; Takenaka et al. 1998), our samples were collected from small clearings under
129 the trees to reduce this effect. It is worth noticing that the Pine trees on Plot 1 and Plot 2 were planted at the end
130 of '60s when most of ¹³⁷Cs fallout had already reached the ground. At that time, the ground was similar to that of
131 the undisturbed area selected as reference site. As a result, in these two plots, fallout interception was minimal
132 during the period covered by the trees and the ¹³⁷Cs inventories are then comparable with those obtained in the
133 reference area (see description below). In Plot 3 and Plot 4, possible fallout interception has occurred due to the
134 longer life of the beech stands (ca. 140 years). However, in Mediterranean areas, most of the rainfall is concentrated
135 in winter time when the deciduous species show no foliage cover. Experimental measurements available for
136 deciduous species (see, for example, Lee 1980; Bruijnzeel and Wiersum 1987) documented values of interception
137 around 10-15% for rainfall higher than 15 mm and for forests with uniform canopy cover. These findings, together
138 with the sampling strategy adopted (consisting of taking samples from small gaps), have probably minimized the
139 effect of fallout interception. However, care was also taken to avoid sampling points close to the tree trunks in
140 order to minimize the effect of stemflow on the final ¹³⁷Cs inventories.

141 In order to obtain information on the ¹³⁷Cs reference value, a detailed soil sampling campaign was also carried out
142 in the reference area located nearby the study plots (elevation = 1400 m a.s.l.). In this case, four separate sectioned
143 cores were collected within a large forest gap, using the same procedure as employed for the sectioned cores

144 collected in the study plots. This sampling strategy was adopted to account for the micro-scale variability of the
 145 atmospheric fallout (see Porto and Walling 2013).

146 After collection, each core was transported in the laboratory and pre-treated for ^{137}Cs measurements. A total of
 147 384 soil samples (336 from the sectioned cores and 48 bulk cores) were collected for the study plots while 120
 148 samples were obtained for the reference area. All the samples were oven dried at 105 °C for 48 h, disaggregated,
 149 dry sieved to separate the <2 mm fraction and packed in plastic pots or petri dishes for determination of its ^{137}Cs
 150 activity by gamma spectroscopy at the University Mediterranea of Reggio Calabria. The gamma-ray spectrometry
 151 equipment used in this study consists of 2 Canberra p-type HPGe detectors, model GX4020. Each detector has
 152 45.6% of relative efficiency with a resolution of 1.1 keV at 122 keV and 2.0 keV at 1.33 MeV and it works coupled
 153 to a Desktop Spectrum Analyzer DSA-1000 Canberra multichannel analyzer. The spectral analysis is performed
 154 using the Canberra Genie 2000 software package. Each detector has been characterized to permit the efficiency
 155 calibration by the Canberra's LabSOCS (Laboratory SOurceless Calibration Software) code that performs
 156 mathematical efficiency calibrations of Ge detectors, without any use of radioactive sources by the laboratory user.
 157 The energy calibration was obtained using a certified multigamma source with a wide energy range (42.8-1274.5
 158 keV). Also, a further validation phase was performed using the activity concentration for ^{137}Cs of several standard
 159 materials of different geometry to cover the specific range of energy of this radionuclide. Count times in the
 160 detectors were typically approximately 80000 s and the ^{137}Cs activities were obtained from the counts at 662 keV.
 161 In order to convert ^{137}Cs measurements into soil erosion and deposition estimates, the diffusion and migration
 162 model (DMM) originally proposed by Walling and He (1999) and refined by Porto et al. (2003), was used. The
 163 DMM simulates the time evolution of ^{137}Cs within a soil profile based on the atmospheric fallout and on its post-
 164 depositional redistribution. Porto et al. (2003) reported the following equation to interpret the activity of ^{137}Cs
 165 within the soil profile (see also Lindstrom and Boersma 1971; Officer and Lynch 1982; Walling and He 1999):

$$\begin{aligned}
 C(x, t, t') = & e^{-\lambda(t-t')} \int_0^{\infty} \frac{I(t')}{H} e^{-\frac{y}{H}} \left\{ e^{\frac{V(x-y)}{2D} - \frac{V^2(t-t')}{4D}} \left[e^{-\frac{(x+y)^2}{4D(t-t')}} + e^{-\frac{(x-y)^2}{4D(t-t')}} \right] \times \right. \\
 & \left. \times \frac{1}{\sqrt{4\pi D(t-t')}} - \frac{V}{2D} e^{\frac{Vx}{D}} \operatorname{erfc} \left[\frac{x+y+V(t-t')}{\sqrt{4D(t-t')}} \right] \right\} dy
 \end{aligned} \tag{1}$$

167 where:

168 $C(x, t, t')$, expressed in Bq kg^{-1} , is the ^{137}Cs activity related to the mass depth x and time t' .

169 D ($\text{kg}^2 \text{m}^{-4} \text{yr}^{-1}$) represents the time-averaged estimate of the diffusion coefficient;
 170 V ($\text{kg m}^{-2} \text{yr}^{-1}$) is the downward migration rate;
 171 H (kg m^{-2}) indicates the relaxation mass depth i.e. the ^{137}Cs initial distribution along the soil profile following the
 172 atmospheric fallout;
 173 λ ($= 0.023 \text{yr}^{-1}$) is the constant of radioactive decay for ^{137}Cs ;
 174 x (kg m^{-2}) is the soil mass depth measured from the soil surface downwards;
 175 t (yr) is the time elapsed since the first fallout;
 176 $I(t')$, expressed in ($\text{Bq m}^{-2} \text{yr}^{-1}$), indicates the ^{137}Cs atmospheric flux at time t' .
 177 $\text{erfc}(u)$ is the error-function complement defined as (Crank 1975):

$$178 \quad \text{erfc}(u) = \frac{2}{\sqrt{\pi}} \int_u^{\infty} e^{-y^2} dy \quad (2)$$

179

180 Taking account of a continuous input $I(t')$, the ^{137}Cs concentration distribution $C(x,t)$ (Bq kg^{-1}) in the soil profile
 181 at time t can be obtained by integrating $C(x,t,t')$ over time t' (Walling and He 1993):

$$182 \quad C(x,t) = \int_0^t C(x,t,t') dt \quad (3)$$

183 The reason of this choice is related to the meaningful results obtained with the use of DMM in southern Italy (see
 184 Porto et al. 2003; 2014) and to the visual inspection of the profiles reported in **Fig. 3** and **Fig. 4** that show a peak
 185 below the soil surface

186

187 2.3 Structural analysis of the study plots

188 To better represent the forest structure in the plots and to correlate soil erosion estimates provided by ^{137}Cs with
 189 the type of forest management in the study area, dendrometric measurements were carried out in the same plots
 190 with the exception of Beech stand 3 where no trees are available. All the trees in the plots were georeferenced and
 191 measurements of height, and diameter have been carried out. Data were then processed to calculate the values of
 192 the Basal Area ($\text{m}^2 \text{ha}^{-1}$), Volume ($\text{m}^3 \text{ha}^{-1}$), Number of plants per plot, and Number of plants per hectare. Moreover,
 193 in order to represent the vertical and horizontal forest structure within the study plots, measures on tree spatial
 194 distribution, canopy insertion height and tree canopy projection on the forest soil were also carried out. Data were

195 then processed through the Stand Visualization System software (USDA, Forest Service) to show the tree spatial
196 distribution for each plot. In addition, hemispherical photos using a fish eye lens were collected and processed
197 with the GLA (Gap Light Analyzer) software for calculating the canopy cover and the leaf area index (LAI).

198

199 **3 Results**

200 3.1 ^{137}Cs inventories and depth distributions at the reference site and study plots and soil erosion estimates

201 The four sectioned cores collected within the reference area produced 120 soil samples (2-cm increment). The
202 matching depth increments were bulked prior to gamma assay, and a representative estimate of the local reference
203 inventory and its depth distribution were obtained. This profile, reported in **Fig. 3** together with the fitting provided
204 by the DMM, is typical of an undisturbed site (see Porto et al. 2003). It shows about 90% of the total inventory
205 occurring in the top 15 cm, a peak in activity ca. 6 cm below the soil surface, and a sharp exponential decline in
206 activity below this depth. This profile shows a reference inventory of 5949 Bq m⁻², in agreement with what found
207 in other areas of South Italy with similar annual rainfall (see Porto and Walling 2012). As mentioned above, the
208 profile was built on the matching depth increments in order to cover the spatial variability of the atmospheric
209 fallout. However, we considered a 10% uncertainty around this value (see Table 1) to account for the measurements
210 error due to the equipment. The values of ^{137}Cs inventory associated with the samples collected from the plots are
211 reported in **Table 1** while the ^{137}Cs depth distributions associated with each plot are illustrated in **Fig. 4**. The four
212 ^{137}Cs depth distributions in **Fig. 4**, one for each plot, indicate different inventory values but they show a similar
213 shape that conforms to that expected for uncultivated soils.

214 Information on the ^{137}Cs inventory values for the study plots, provided in **Table 1**, indicates that 29 out of 36
215 sampling sites (including the profiles in **Fig. 4**) show inventory values lower than the reference value pointing out
216 evidence of soil erosion within the plots. Only 7 points documented evidence of deposition with ^{137}Cs inventories
217 higher than the reference value. A mean value of 4689 Bq m⁻² (lower than the reference inventory), obtained from
218 the 36 single points, showed that erosion was the dominant process during the time span (ca. 60 years) covered by
219 the ^{137}Cs measurements.

220 Information on the corresponding estimates of soil loss provided by the DMM is also reported in **Table 1** for each
221 plot.

222

223 3.2 Structural data of the study plots

224 Data obtained from the structural surveys are listed in **Table 2**. Pine stand 1, Pine stand 2, and Beech stand 4
225 showed respectively a mean diameter of 32.31 cm, 33.27 cm and 36 cm. The same stands showed volume values
226 of 699 m³ ha⁻¹, 1589 m³ ha⁻¹ and 1080 m³ ha⁻¹ while the number of trees per hectare was 866, 1500 and 933,
227 respectively. The tree canopy cover determined with the use of hemispherical photos and the GLA software was
228 52% for Pine stand 1, 63% for Pine stand 2 and 51% for Beech stand 4. The corresponding values of the leaf area
229 index (LAI) were 0.79 (m² m⁻²), 1.13 (m² m⁻²) and 0.80 (m² m⁻²), respectively.

230

231 4 Discussion

232 The results reported in **Table 1** show different values of ¹³⁷Cs inventories. The significance of these differences
233 was checked, preliminarily, using the Friedman test (see Kanji 1993). The resulting statistics were always higher
234 than the critical value at the 0.05 level of significance, indicating that there were significant difference between
235 the 4 empirical distributions of ¹³⁷Cs values associated with the plots.

236 The ¹³⁷Cs measurements carried out in the 9 sampling points selected in each plot provided an estimate of the
237 average soil redistribution rates for the period extending from the commencement of ¹³⁷Cs fallout in the mid-1950s
238 to the present. As explained above, the use of these measurements was based on the comparison between the
239 radionuclide reference value obtained in the reference area and the ¹³⁷Cs inventories measured in the 9 single points
240 for each plot. Most of the latter inventories (29 out of 36) were lower than the reference value, indicating that soil
241 erosion was a dominant process during the last ca. 60 years. The mean value of the erosion rates, reported in **Tab.**
242 **1** for each plot, range from 0.38 t ha⁻¹ yr⁻¹ to 1.34 t ha⁻¹ yr⁻¹. These values are generally lower than the annual mean
243 values considered as ‘issue of concern’ by the scientific literature. Verheijen et al. (1999), for example, even when
244 dust deposition is included in soil formation rates, reported an upper limit of tolerable soil erosion of ca. 1.4 t ha⁻¹
245 yr⁻¹. These findings support strongly the hypothesis that forests play a key role in regulating the runoff and soil
246 erosion, due to the protection offered by the canopy cover and to their capability of improving the absorption
247 capacity of soils. In fact, both these functions are useful to reduce the overland flow and the water erosion risks
248 related. However, some differences between the four forest types selected in this study must be recognised. In
249 order to facilitate the comparison between the four treatments, a spatial visualization of the trees, obtained with
250 the SVS software, is provided in **Fig. 5A-D**. The corresponding spatial distribution of soil erosion rates was also
251 created, using a kriging interpolation procedure, and is depicted in **Fig. 5E-H**.

252 A first comparison can be made between Plot 1 and Plot 2, having the same forest type (Pine stand) but with
253 different forest cover and biomass. Pine stand 1, characterised by a canopy cover of 52%, shows a mean value of
254 soil erosion equal to $1.16 \text{ t ha}^{-1} \text{ yr}^{-1}$ that is higher than that documented by Pine stand 2 (only $0.26 \text{ t ha}^{-1} \text{ yr}^{-1}$) where
255 the canopy cover is 63%. The higher value of soil erosion in Pine stand 1 could also be due to a minor tree density
256 ($866 \text{ trees ha}^{-1}$) compared to that ($1500 \text{ trees ha}^{-1}$) measured in Pine stand 2 (see also Fig. 5_{A,B}). A higher tree
257 density in fact, enhances the soil protection due, for example, to a stronger root system and a larger surface taken
258 up by the tree canopy that intercepts a higher amount of rainfall and limits the raindrop impact on the ground (see
259 Chapman 1948). The different forest covers and tree density currently found in both pine stands are part of the
260 same type of silvicultural treatment. These differences are probably due to micro-variability (e.g. small disturbances
261 and stand dynamic evolution) in that area (Gray and Spies 1996). Anyway, the management of the stand in the
262 past years preserved the protective role of the trees against soil erosion processes. These findings emphasise the
263 conclusion that an increase in forest cover and biomass improves soil protection. The pattern of soil erosion rates
264 depicted in Fig. 5_{E,F} clearly delineates the areas with the highest rates of soil loss within the plots. A close
265 correspondence with the lower values of canopy cover can be observed (see Fig. 5_{A,B}) and this supports the above
266 conclusion.

267 A second important comparison can be observed between Plot 1 and Plot 4 having similar canopy cover (52% and
268 51%) but characterised by two different forest species (Pine and Beech, respectively). The mean value of soil
269 erosion in Beech stand 4 ($0.68 \text{ t ha}^{-1} \text{ yr}^{-1}$) is lower than that documented in Pine stand 1 and seems to suggest a
270 better protection against soil erosion offered by the former. A possible explanation of this result can be related to
271 the greater canopy cover capacity expected for deciduous species than for conifers, as documented by other authors
272 (see among the others, Helvey and Patric 1965; Zinke 1967). Another possible reason for these lower values of
273 soil erosion rates can be ascribed to the Beech root system that is generally shallow and superficial, with large
274 roots spreading out in all directions. Conversely, the Pine root system is relatively deeper and does not offer the
275 same protection against erosion of the first soil layers. However, the role of the canopy cover in protecting the soil
276 from erosion is confirmed by the spatial pattern of soil erosion rates depicted in Fig. 5_H. In fact, a comparison with
277 the spatial visualization of the trees in Fig. 5_D shows, again, a close correspondence of the areas with higher erosion
278 rate with the lower values of canopy cover.

279 A last important comparison can be made between the Beech stand in Plot 4 and the forest system in Plot 3,
280 originally covered by a Beech stand that was clearcut in 2009. In this case, it was possible to explore the effect
281 reflected by the clearcutting in terms of soil loss from this area. The mean value of soil erosion rate for the Beech

282 stand 3 is $1.34 \text{ t ha}^{-1} \text{ yr}^{-1}$, almost double that from Beech stand 4. This result is clearly not unexpected because the
283 temporary lack of canopy cover occurred during the last 7 years may have affected soil properties and this resulted
284 in significant increase of soil loss in recent periods. However, it must be recognised that the last 6-7 years following
285 the clearcutting are characterised by the occurrence of very severe rainfall events that coincided with the absence
286 of vegetation in this area and may have affected strongly the rate of soil loss in the same period. Although no direct
287 measurements of soil erosion are available for the recent years, an attempt to reconstruct the trend of soil loss using
288 the rainfall erosivity factor (Wischmeier and Smith 1978) can be made. This trend is depicted in **Fig. 6** for
289 Gambarie meteo station located in the vicinity of the study site. The dataset of the 30-min rainfall is available from
290 1991 to date. The clearcutting occurred in October 2009 and **Fig. 6** indicate that for the set of storm events occurred
291 from that period to the sampling date, the mean value of the R-factor ($R_m = 4473 \text{ MJ ha}^{-1} \text{ mm}^{-1} \text{ h}^{-1}$) was significantly
292 greater than that related to the period 1991-2008 ($R_m = 2635 \text{ MJ ha}^{-1} \text{ mm}^{-1} \text{ h}^{-1}$). The increase is particularly marked
293 for the events that occurred in 2015, prior to the sampling campaign. Similar trends of the R-factor, with a general
294 increase during the last decade, have been also documented by Capra et al. (2017) for other stations located in the
295 same region. The higher values of the R-factor, combined with the tree cutting showed a general increase of soil
296 erosion even in other areas, emphasizing the importance of vegetation in protecting the soil (see Porto et al. 2009).
297 However, further measurements are needed during the next years to establish if the effect of the clearcutting might
298 be mitigated while the new vegetation will take place.

299

300 **5 Conclusions**

301 The overall results obtained in this experiment demonstrated, first of all, the ability of ^{137}Cs as a tracer technique
302 to predict soil erosion rates. Based on the ^{137}Cs measurements, minimum values of soil loss ($0.26 \text{ t ha yr}^{-1}$) are
303 documented in areas covered by Pine stand with higher canopy cover and biomass (Plot 2), evidencing the
304 protective effect provided by forests against soil erosion. The experiment demonstrated that when canopy cover is
305 lower and discontinuous, as in the two cases described in Plot 1 and Plot 4, the ^{137}Cs measurements documented
306 higher values of soil loss, equal to $1.16 \text{ t ha}^{-1} \text{ yr}^{-1}$, and $0.68 \text{ t ha}^{-1} \text{ yr}^{-1}$, respectively. These findings indicate that
307 when afforestation programmes or new plantations activities are planned, but also during the silvicultural
308 treatments, it is necessary to reduce the occurrence of large gaps among the forest cover in order to reduce soil
309 erosion. The effectiveness of canopy is also evident in the uncovered area (Plot 3) where the original Beech stand
310 was clearcut in 2009. In this case, the mean rate of soil loss provided by the ^{137}Cs measurements was 1.34 t ha^{-1}

311 yr⁻¹, almost double that in the area (Plot 4) where the same forest type was maintained undisturbed. Although the
312 clearcutting operations occurred in a period during which the rainfall erosivity was the highest in the last 25 years,
313 the experiment indicates that care must be taken when new silviculture treatments are planned. These findings are
314 in agreement with what documented by other authors in similar environments but further work is necessary to
315 establish the effectiveness of ¹³⁷Cs measurements in different forest contexts.

316

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320

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- 436
- 437

438 Table 1. ¹³⁷Cs inventories for the reference area and the study plots, and calculated values of soil redistribution
 439 rates (negative values indicate erosion).

	Reference inventories		Inventories within the plots			Soil redistribution rates	
	Mean (Bq m ⁻²)	Range (10% uncertainty) (Bq m ⁻²)	Mean (Bq m ⁻²)	Range (Bq m ⁻²)	St. Dev. (Bq m ⁻²)	Mean (t ha ⁻¹ yr ⁻¹)	Range (t ha ⁻¹ yr ⁻¹)
Pine stand 1	5949	5354-6544	4185	1897- 10943	2853	-1.20	-2.6 – 3.16
Pine stand 2			5609	3309- 10211	2147	-0.26	-1.7 – 2.68
Beech stand 3			3910	1374-6594	1586	-1.34	-3.0 – 0.41
Beech stand 4			4910	2845-7267	1399	-0.68	-2.0 – 0.84

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442

443 Table 2. Dendrometric values obtained for each plot

Plot	Mean DBH (cm)	Mean H (m)	Basimetric area (m ² ha ⁻¹)	Volume (m ³ ha ⁻¹)	No. of trees per hectare	Canopy cover (%)	LAI (m ² m ⁻²)
Pine stand 1	32.3	19.9	74.1	698.9	867	52	0.79
Pine stand 2	33.3	24.7	138.9	1589.9	1500	63	1.13
Beech stand 4	36.1	22.8	102.6	1080.4	933	51	0.80

444

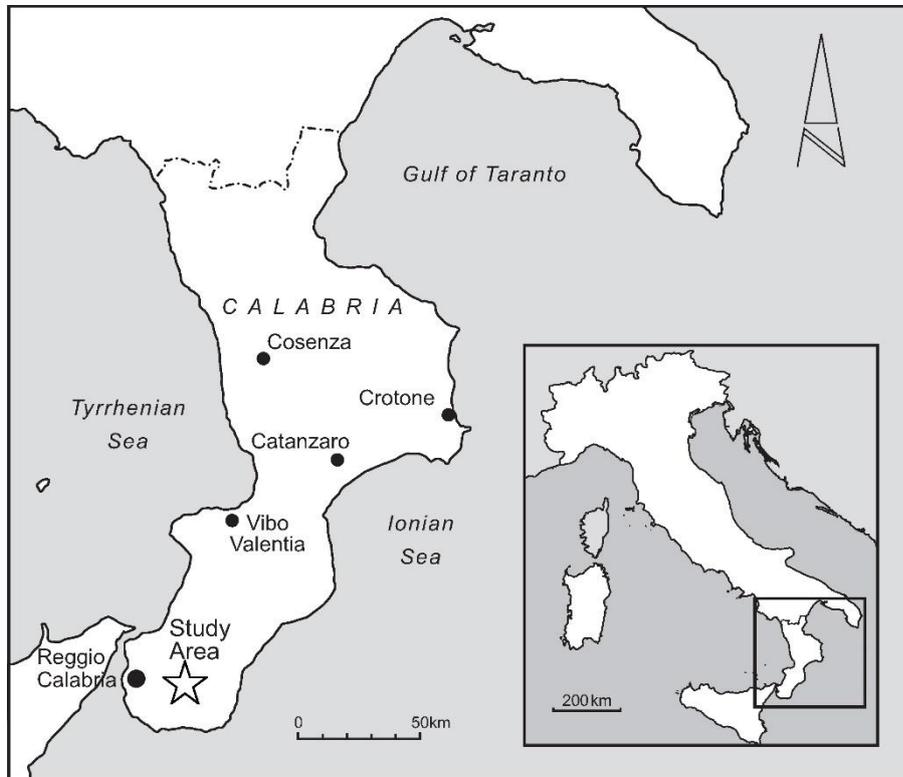
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447 **Figure**

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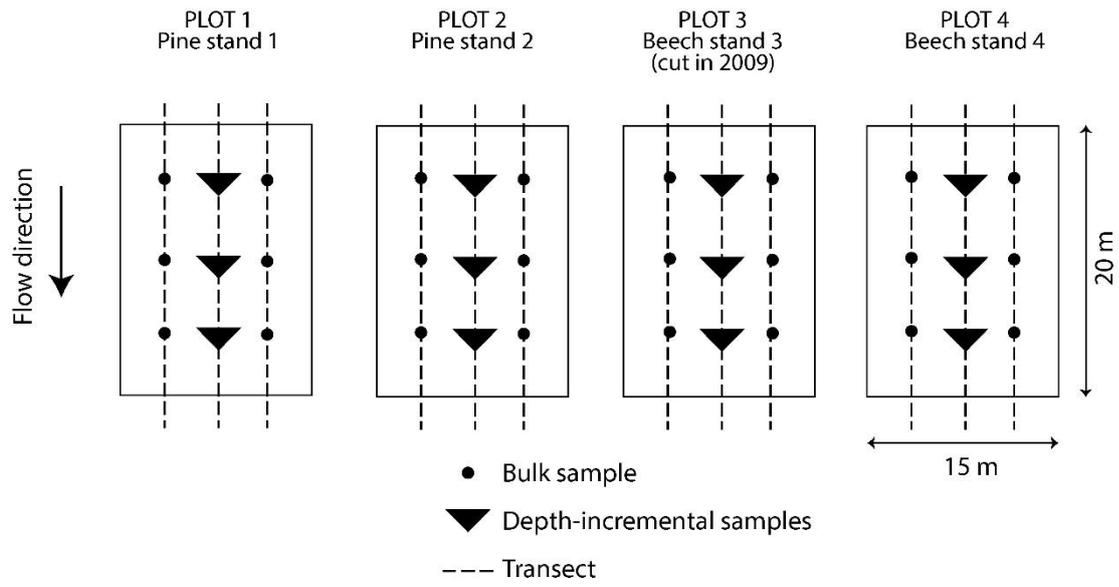
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451 **Fig. 1** The study area

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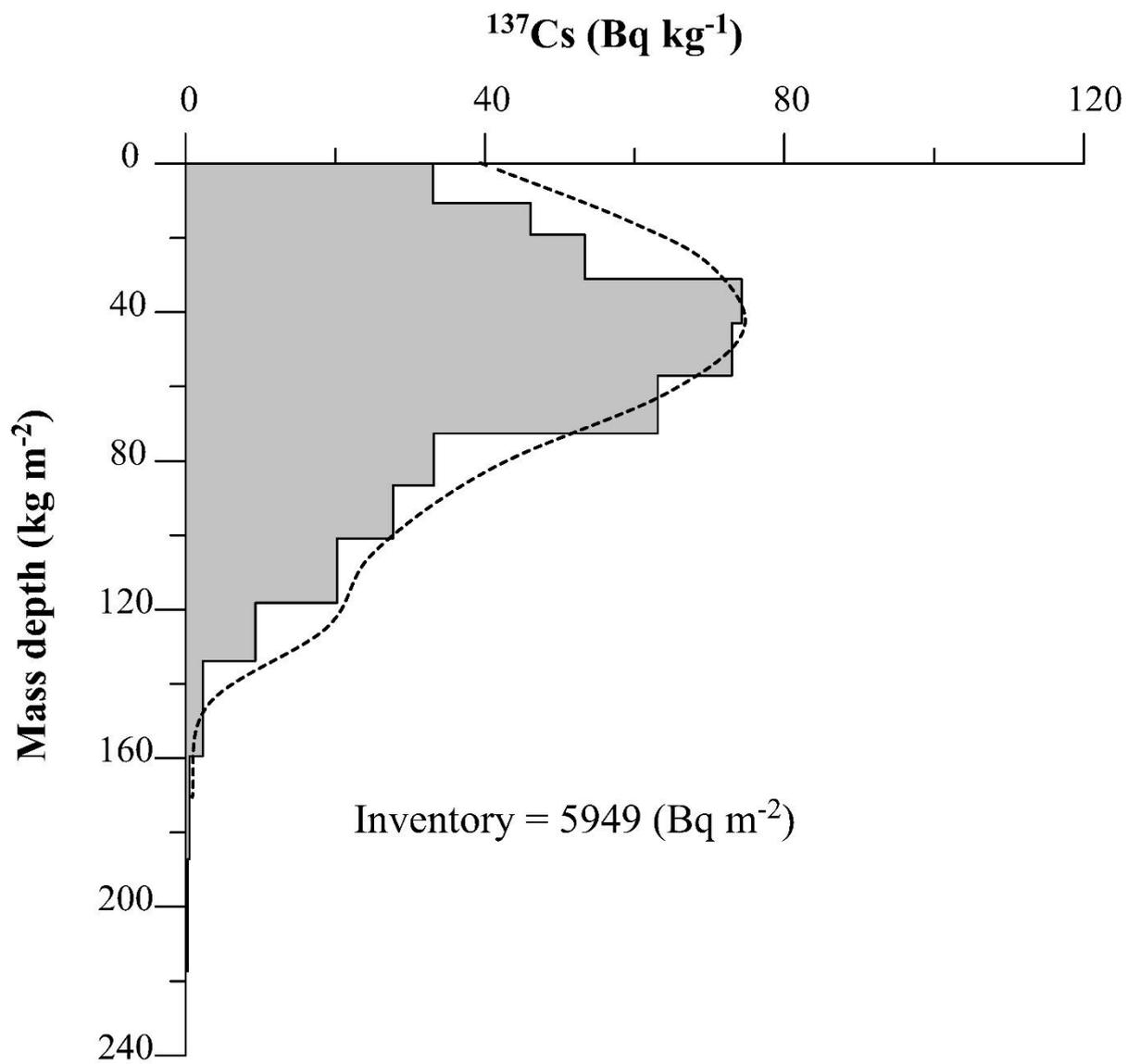
456 **Fig. 2** The experimental plots and the sampling locations

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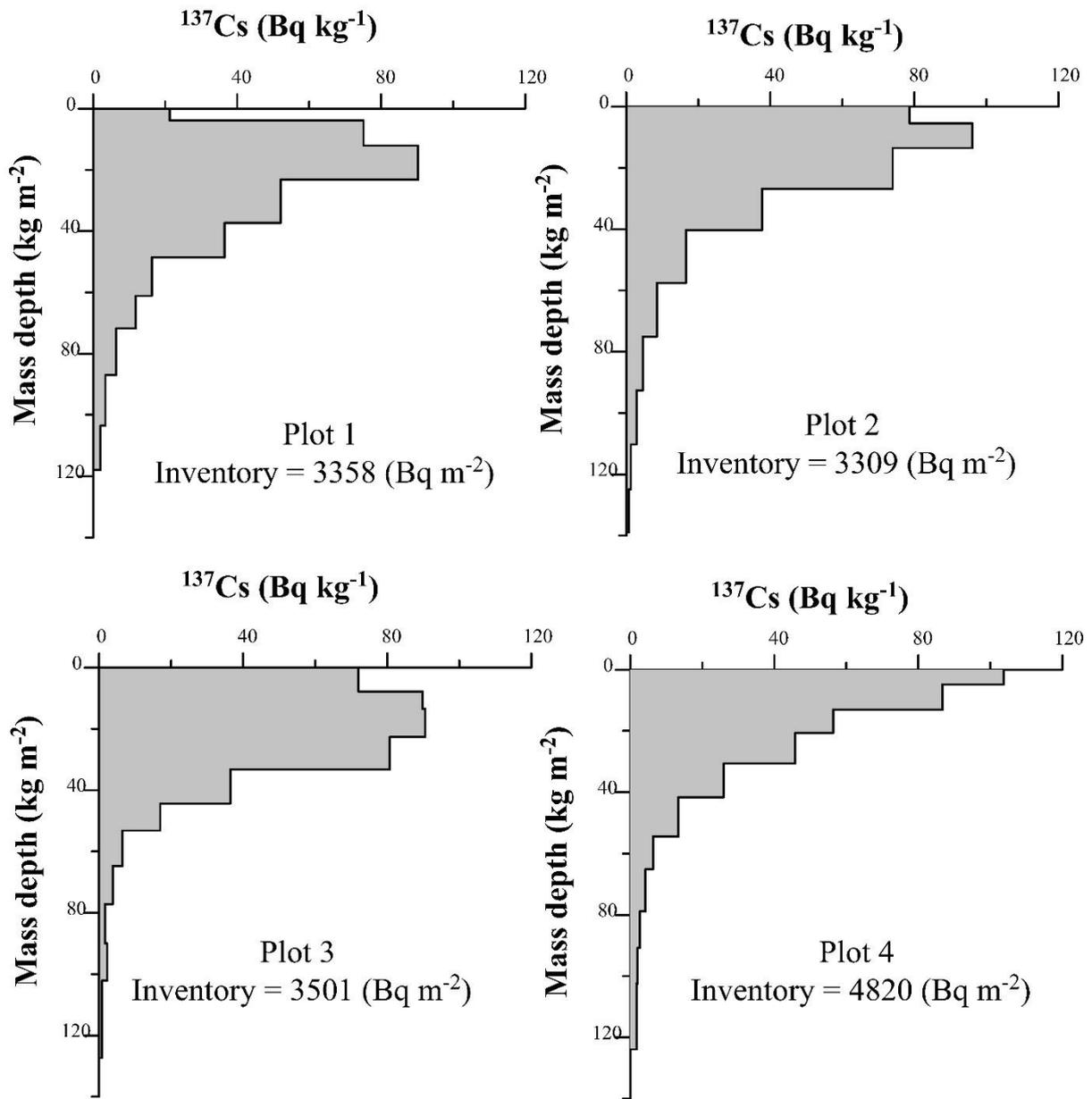
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462 **Fig. 3** ^{137}Cs inventory and depth distribution in the reference area

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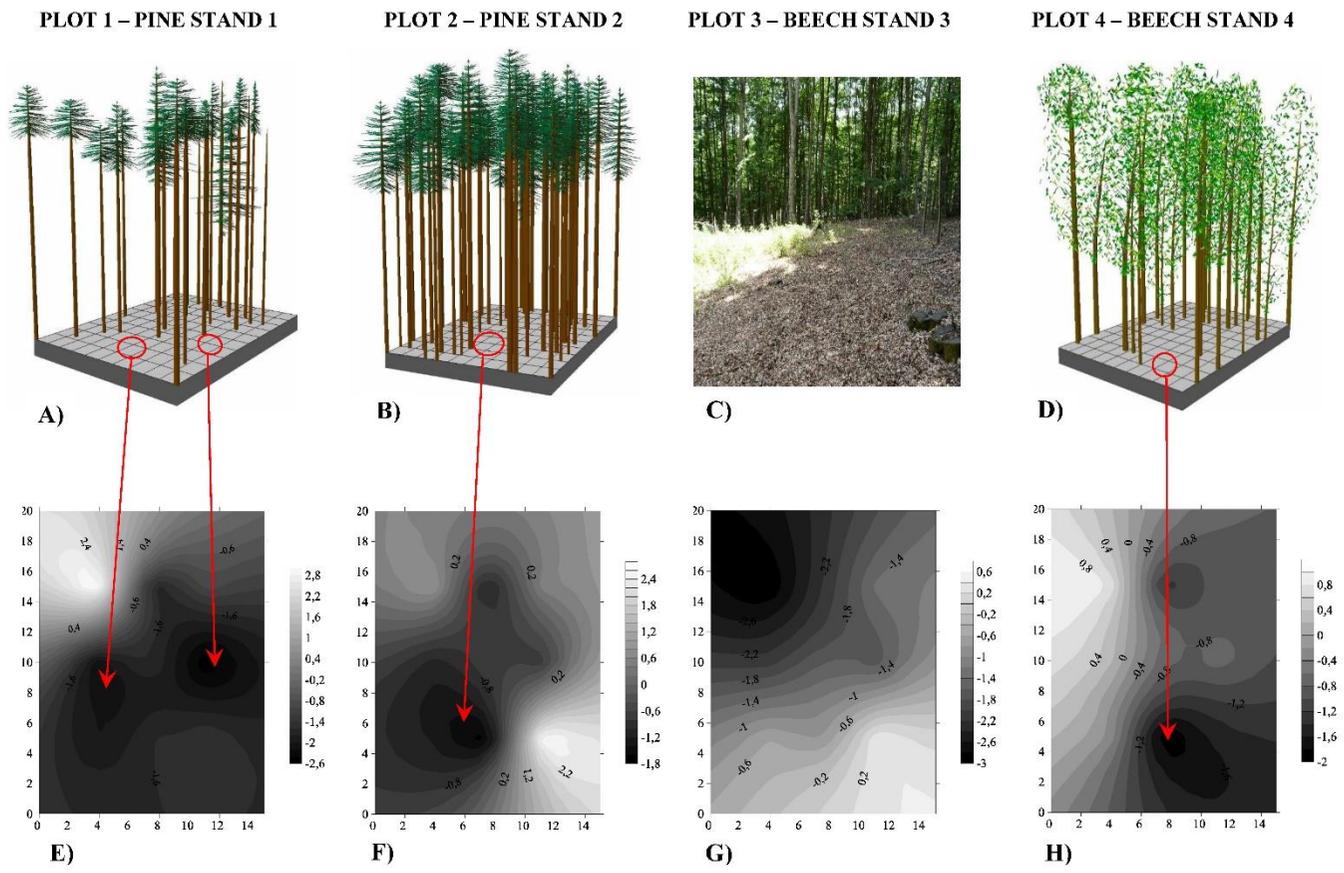
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467 **Fig. 4** ¹³⁷Cs inventory and depth distribution within the study plots

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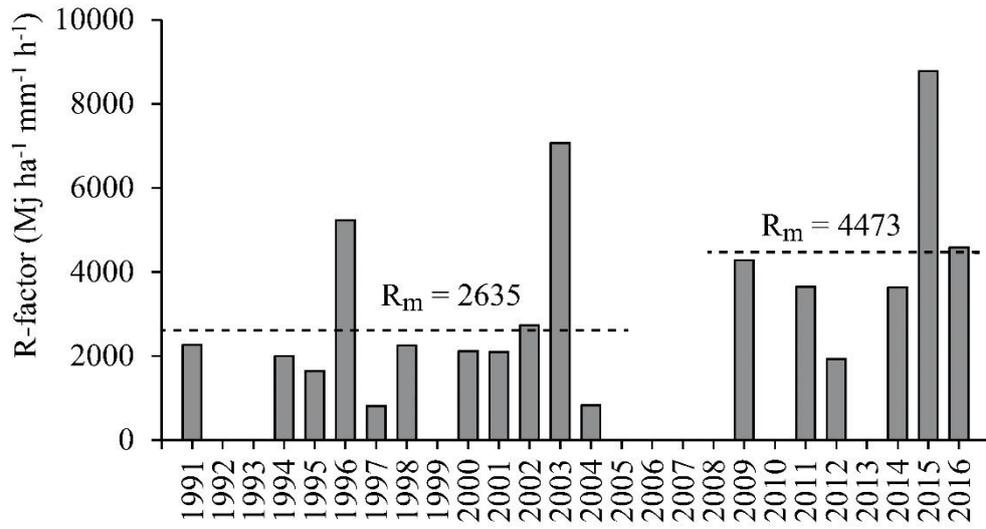


471

472 **Fig. 5** Spatial vision of the trees from each plot (A-D) and corresponding spatial distribution of soil erosion (E-H). Figure 5C shows a photo of the Plot 3 where cutting
 473 operations were performed in 2009.

474

Gambarie meteo station



475

476 **Fig. 6** Rainfall erosivity factor for Gambarie meteo station

477