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Silvicultural systems effects in soil erosion processes in mountainous Mediterranean forests

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Short abstract

Forests play a key role in providing protection against soil erosion. In this work, two methods were used to evaluate soil erosion in relation to different forest management, in Mediterranean forests. ^{137}Cs technique is linked to the global fallout of bomb-derived radiocaesium (1950s-1970s) and is based on a comparison between ^{137}Cs inventory measured at the study site and that obtained at a reference site. RUSLE2 model is a software program which makes soil erosion estimates based on site specific conditions. Here, four study plots in the southern Apennines (Italy) were selected in pine and beech high forests located in Aspromonte site. The measurements related to forest structural characteristics were first compared with soil erosion rates obtained with ^{137}Cs technique application. Secondly the climate, soil, topography and management information belonging to the areas previously selected, were used to make soil erosion estimates under the RUSLE2 model. The ^{137}Cs results revealed that in areas with higher biomass and canopy cover the soil loss values were low, highlighting the protective function of forest stands against soil erosion. Meanwhile, the soil loss values provided by the RUSLE2 assumptions, were shown to be affected by different aboveground biomass (ABG) amounts (higher soil loss values in areas with low amounts of ABG) and disturbance brought by silvicultural operations. The two methods were proved to be useful to predict soil erosion rates, demonstrating the importance of canopy cover and aboveground biomass and pointing out the need of planning adequately forest management options. However, further information and runs in RUSLE2 are needed to validate the model assumptions and extend its range of application as well as additional ^{137}Cs measurements are required in order to enhance the accuracy of the estimates in different forest environments, in a long-term perspective.

Keywords: Cesium-137, Apennine forests, RUSLE2 model, soil erosion rates, protective role.

Extended abstract

1 Introduction

This PhD thesis deals with the study of the role of forest stands and their management on mitigating soil erosion processes in Mediterranean environment. Particularly, the role of the vertical stratification in forests is crucial for the rainfall interception, in order to mitigate the direct impact of water with soil. In this work, two analytical methods were used to evaluate soil erosion in relation to different forest management, in Mediterranean mountain forest: ^{137}Cs technique and RUSLE2 computer program.

1.1 ^{137}Cs technique applications

Caesium-137 is a human-induced environmental radionuclide, released into the atmosphere after nuclear weapons tests occurred in the timeframe extending from 1950s to 1970s. The strong association between Cs and soil has made it a suitable method for dating and tracing purposes (Owens et al., 1997. Moreover, the comparing between net erosion measurements on a long term and the estimate derived from this technique has led to interesting results for soil with forest cover (Porto et al., 2004) as well as in this case study. Despite the increasing application of this technique, there is still a lack in the research about some feasible relationships between the different types of forest management and the soil loss rate obtained in a forest environment.

1.2 The RUSLE2 model

As regard the RUSLE2 model, various prediction models and procedures have been implemented to select management goals and strategies, including mitigating forest soil erosion as part of a sustainable forest management plan. The USLE/RUSLE/RUSLE2 approach (USDA-ARS 2013; Renard et al., 1997) is one of the most commonly used of the models. In Italy, for example, Napoli et al. (2016) compared scenarios of soil erosion by means of GIS at a regional scale for application to the RUSLE model. Most of these works have been for standard row-crop agriculture, but managing so as to reduce erosion is also a concern for forested areas. Early on the USLE was adopted for use in these situations (Dissmeyer and Foster, 1981), and that approach is still useful for understanding the cause and effect relationships between management practices and erosion. The RUSLE2 (Revised Universal Soil Loss Equation, Version2) model makes estimates based on specific site conditions. It allows erosion control practices to be tailored to each specific site, and tries to make the estimate as independent of land use as possible (Dabney et al., 2011).

2 Materials and Methods

2.1 Study area and field surveys

In this work two-field campaign were undertaken in Ferraina site (Aspromonte National Park). During the first fieldwork phase, four plots and a reference site were selected in beech and pine stands. At each of these plots, three separate soil cores were collected, while four cores in the reference site were collected. In parallel, measurements of forest stand structure. The second fieldwork phase was carried out to collect six additional soil cores for each plot. The trees included in the plots were measured (height and diameter) to calculate dendrometric measurements. The Stand Visualization System (SVS) software (USDA, Forest Service) was then used to show the tree spatial distribution for each plot. In addition, hemispherical photos using a fish eye lens were collected and processed with the GLA (Gap Light Analyzer) software (2.0 version, Simon Fraser University, Canada) for calculating the canopy cover and the leaf area index (LAI).

2.2 ^{137}Cs measurements

The laboratory activities and data processing were carried out at University 'Mediterranea of Reggio Calabria'. A total number of 384 and 120 soil samples were analysed for the study plots and the reference site respectively. Soil samples were then analysed for determination of its ^{137}Cs activity by gamma spectroscopy in the radiometric laboratory. ^{137}Cs measurements were converted into soil and deposition rates through the application of the diffusion and migration model (Walling and He, 1999).

2.3 Particle size analysis

A standard procedure for preparing soil samples for absolute particle size analysis was followed (Walling & Collins, 2000). Subsequently, the size distribution of the samples was determined using a size analyser named ‘Laser Particle Sizer analysette 22’. PSA data can be shown in several ways, for example being a cumulative particle size distribution curve, usually exhibiting one or more peaks representing the most prevalent particle sizes.

2.2 RUSLE2 application

The parameters of *Climate*, *Soil*, *Topography* and *Management* description of the stands was set. This information was required by RUSLE2 model in order to address all the RUSLE2 factors in (1) and make the estimate of the long-term erosion and compare to that measured on both pine and beech plots using ^{137}Cs techniques.

$$A = R * K * LS * C * P \qquad (1)$$

In order to run data in RUSLE2 model, based on the same study plots where the ^{137}Cs technique was applied, the following descriptions were defined: rainfall erosivity description as well as the long-term average monthly temperature and precipitation values (*Climate*); soil texture to derive ‘K soil erodibility’ values (*Soil*); specification of length by modelling plots as single uniform segments (20 meters), and steepness for pine and beech respectively (*Topography*); examination of the five RUSLE2 C dimensionless subfactors (2) and modelling of the annual addition and decomposition of surface litter as it follows the annual canopy increase and/or decrease, to evaluate the impact of forests on erosion (*Management*).

3. Results and discussion

3.1 ^{137}Cs and dendrometric measurements

Based on a comparison between the radionuclide reference value obtained in the reference area and the ^{137}Cs inventories measured, results about the latter plots highlighted inventory values lower than the reference value. This indicates that the depth distribution is typical of an undisturbed site for the reference area and that soil erosion was a dominant process during the last ca.60 years. Some of these results are reported in the table below (Table 1). Differences were also found through the comparison of the plots, for example between Plot 1 and Plot 2 having the same forest type (Pine). In this case, the higher value of soil erosion could be due to lower canopy cover or a minor tree density in Pine stand 1, compared to that measured in Pine stand 2. Higher values of tree density or canopy cover in fact, can improve the soil protection due to a larger surface taken up by the canopy that intercept a higher amount of rainfall (Chapman, 1948).

Table 1 ^{137}Cs inventories in the reference area and the study plots.

Inventories	Reference area		Study plots			Net erosion
Plots	Mean (Bq m^{-2})	Range (10% uncertainty) (Bq m^{-2})	Mean (Bq m^{-2})	Range (Bq m^{-2})	Standard deviation (Bq m^{-2})	Mean (Mg ha^{-1} yr^{-1})
Pine stand 1			4185	1897-10943	2853	1.16
Pine stand 2	5949	5354-6544	5609	3309-10211	2147	0.26
Beech stand 3			3910	1374-6594	1589	1.34
Beech stand 4			4910	2845-7267	1399	0.68

3.2 Determination of soil texture

The measurements produced the particle size curve of each single sample. According to soil texture diagram, the soil characteristics related to the four plots, gave a silt loam texture. This reveals appropriate amounts of organic matter and nutritive elements, commonly observed in forest soils, just like the one of the conifer species.

3.3 RUSLE2 runs

The main outcome on RUSLE2 runs show soil loss values for each plot ranging from 2.3 to 0.24 Mg ha^{-1} as reported in Table 3 a-b. For Pine Stand 1 and Pine Stand 2 the *Management* descriptions began with existing RUSLE2 descriptions of relatively low-productivity (7 Mg ha^{-1} yr^{-1}) forage, representing the assumed 21 years (1954-1975) of poor-quality pasture followed by planting of the pine plantation and its growth until 2016. The sole difference between the two pine stands is in the modelling of the *Vegetations*, with the Pine Stand 1 *Vegetation* modelled to yield an average surface litter mass equal to the measured 374 m^3 ha^{-1}, while the Pine Stand 2 *Vegetation* matched the measured 798 m^3 ha^{-1}. The Beech Stand 3 *Management* was modelled as 56 years of beech followed by 7 years of grass, while Beech Stand 4 was modelled as 63 years of beech trees. In both cases the *Vegetations* were designed to yield the measured litter masses, being 1160 m^3 ha^{-1} for Beech Stands 3 and 4. On reaching maturity, the pine plots aboveground biomass stabilized at about 374 and 798 m^3 ha^{-1} for plot 1 and 2 respectively.

Table 3 Estimated results for RUSLE2 climate, soil, vegetation and management inputs in Pine stand plots (a) and Beech stand plots (b).

Management			Soil loss ($Mg\ ha^{-1}\ yr^{-1}$)		
Plots	Operations	Vegetation	Mean	St. Dev.	Variability %
Pine stand 1	Begin new growth Basic/general/kill vegetation Begin new growth	New forages/Upper Mid South/Tall Fescue with Bermudagrass. Conifer from planting	2.3	0.45	19
Pine stand 2	Begin new growth Basic/general/kill vegetation Begin new growth	New forages/Upper Mid South/Tall Fescue with Bermudagrass. Conifer from planting	0.24	0.080	33

a)

b)

Management			Soil loss ($Mg\ ha^{-1}\ yr^{-1}$)		
Plots	Operations	Vegetation	Mean	St. Dev.	Variability %
Beech stand 3 (cut in 2009)	Begin new growth	Established forest, beech	0.47	0.19	41
Beech stand 4	Begin new growth	Established forest, beech	0.40		

4. Conclusions and Future perspectives

Among the several methods to estimate the phenomenon of soil erosion process, ^{137}Cs technique was confirmed in this study as a valid methodology to be applied in a forest environment. The high values of soil loss found (1.16 $Mg\ ha^{-1}\ yr^{-1}$ and 0.68 $Mg\ ha^{-1}\ yr^{-1}$ for Pine stand 1 and Beech stand 4, respectively) can be linked to a low and discontinuous canopy cover or to a different forest treatment in the study area. This proves that the afforestation programmes and silvicultural operations (even if applied in low-intensity) must be planned carefully. One of the attempts in this work was to provide an experimental comparison of ^{137}Cs method with other methodologies to assess erosion rates, which was carried out using RUSLE2 model. The results of the runs showed erosion rates of 2.3 and 0.24 $Mg\ ha^{-1}yr^{-1}$ for the pine plots 1 and 2, respectively. These values reflect the different biomass amounts used for the runs. These amounts, belonging to forest stands, include different vegetal components compared to the usually considered in the software, namely croplands, whose amounts are far more limited than those in forest. Therefore, the high sensitivity observed from the sensitivity analysis to the biomass parameter confirms that the model is not calibrated for large amounts of biomass.

Since this model finds its range of application in cropland systems, further trials on several forest study areas could improve the functionality concerning these issues, in order to extend the range of applicability to forested areas. The RUSLE erosion estimates for the beech plot resulted different. Under the steady-state mature forest assumption in fact, the estimated erosion of 0.45 Mg $ha^{-1}yr^{-1}$ was far below the ^{137}Cs value of 0,68 Mg ha^{-1} yr^{-1}.However, further information on management and runs in the model are needed to validate RUSLE assumptions as well as additional ^{137}Cs measurements are required in different forest environments.

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1 Introduction and state of the art

Forests are the most predominant ecosystem on the Earth (Pan et al., 2013), covering approximately 30% of the total land area (FAO, 2010). They harbour most of the living organisms and provide goods and services to humanity, including timber, food, water and climate moderation (Jackson et al. 2005, McKinley et al. 2011). During the past century, most of the forest cover has been cleared because of the deforestation, which is estimated to be 13 million hectares each year. This process of destruction linked to fragmentation and land degradation has been triggered by human activities like unsustainable forest management, overgrazing, conversion of forests to agricultural land, forest fires, pollution and climate change, causing a concerning decrease of biological diversity (Balmford et al., 2005).

One of the several consequences of these activities is the soil degradation, represented by the displacement of the upper layer of soil, in which wind and water are the primary causes. This process, naturally occurring on all land at low levels, can become alarming when stimulated and altered by anthropogenic activities (Julien, 2010). FAO (2010) studied soil degradation as a process, which decreases the potential capability of the soil to produce goods and services. Among the several land categories, forests and woodland intended as land under natural or planted stands of trees and forests cleared, are affected by this process, worsened by human induced actions like deforestation and overexploitation of the vegetative cover (Oldeman, 1992). In mountainous areas, forests are important for several ecological functions and for providing environmental services and benefits. Its multiple function has gained attention over the years so much to emphasize the cultural, social and ecological service function complex (Hamilton, 1992).

Over the last decades, the growing international attention towards this issue has been widely acknowledged by many organizations and conferences like 'UN Conference on Environment and Development (UNCED)' in Rio de Janeiro in June 1992 and 'Strasbourg European ministerial conference' (Motta and Haudemand, 2000). The implementation of the 'Organic Administration Act of 1897' in the United States provided the basis for the management of forest reserves intended as systems for multiple use and gave priority to the aim to enhance and preserve forests, water flows and timber extraction (Sakals et al, 2006). On ecological scale, forested areas supply the oxygen essential to survive, play the role of habitats for plant community, animal species, fungi and bacteria. On a productive scale, they can provide fuelwood and livestock feed or be a valid backdrop for tourism (Hamilton, L. S. 1992). Among all the forest functions, the protective is one of the most important and it is performed against landslides, avalanches, rock falls and debris flows. According to the third Ministerial Conference for the Protection of Forests in Europe (MCPFE - Lisbon, 1998), the forest management includes both direct and indirect protection: the first one aims to preserve human life and activities from natural hazards while the second one regulates the water flow and prevents soil erosion (Motta and Haudemand, 2000).

The protective role of forest against soil erosion is considered as a relevant phenomenon linked to pedogenesis and natural ecosystems evolution. Factors such as climate, organisms, biological processes, and time are closely related to determine soil formation (Jenny, 1981) and promote its soil binding, aeration and water hold capacity.

Considering water erosion as one of the most widespread forms of degradation occurring in woodland areas, forest vegetation is important in the interception of the precipitation, in order to mitigate the direct impact of water with soil. The mitigation effect is the result of rainfalls partitioning determined by the vegetation structure (Herwitz, 1985). According to different forest types and climates the range of throughfall can vary widely: from 70 to 90% of rainfall in broadleaf and coniferous temperate forests and from 60 to 95% of rainfall in tropical forests (Berger et al., 2008; Bruijnzeel, 2004; Crockford and Richardson, 2000; Germer et al., 2006; Levia and Frost, 2006; Llorens and Domingo, 2007; Lloyd and Marques, 1988; McJannet et al., 2007b; Vernimmen et al., 2007; Zimmermann et al., 2007). Rainfall interception and soil loss are also linked to other ecological factors like vegetative cover, forest community structure (Kimmins, 1987), soil structure and composition, climate and topography. Soil loss phenomenon is included in the wider context of soil degradation that is defined as "the process of the hydrological, biological, chemical, and physical properties changes of the soil" (Schoenholtz et al., 2000). This process is considered as one of the main effects contributing to the alteration of the ecosystem functions (Kirkby et al., 2000). The main causes of these severe erosive phenomena are attributed to extreme meteoric events (Smith et al., 2011; Shakesby, 2011), management practices not always correct (Cerdà and Lasanta, 2005) and particular morphological land conditions (Bouma, 1989). Several best management practices, for example, are used in forestry operations to mitigate the impact of logging on stream water quality. These practices are designed to control and decrease erosion phenomena and to minimise the delivery of sediment and pollutants off-site to natural drainage lines (Wallbrink and Croke, 2002).

The need to make easier management options on land use induced wide discussions of this aspect in most studies all over the globe. Studies on water erosion were made in Japan, requiring investigation of the spatial variability of factors affecting the erosion processes (Yang et al., 2003) and using common physical parameters i.e. precipitation, slope, soil erodibility and vegetation cover within the models (Siakeu and Oguchi, 2000). In the United States, soil erosion data have been processed on a long-term period by using standardized methodologies linked to the Universal Soil Loss Equation (USLE) and the (Revised) Universal Loss Equation (Wischmeier and Smith, 1978; Nearing et al., 2000). Besides, additional models and several calculation procedures including climate, topography, soil and vegetation parameters have been realised and largely used for estimating purposes, in order to prevent this phenomenon: European Soil Erosion Model (EUROSEM) (Morgan et al., 1992), LImburg Soil Erosion Model (LISEM) (De Roo *et al.*, 1996), Water Erosion Prediction Project (WEPP) (Nearing et al., 1989). In Europe, most of the data come from bounded field plots under natural rainfall roughly comparable to those estimated by the United States Department of Agriculture (USDA) (Cerdan et al., 2010).

Because of its prolonged dry periods followed by severe rainfalls occurring on steep slopes and fragile soils, the Mediterranean basin is one of the many areas affected by erosive events (Grimm et al., 2003).

In most Mediterranean lands in fact, these events are a serious risk for land degradation and desertification, declining vegetation growth, siltation of water courses and the formation of deltas along coastal areas (Kosmas et al., 1997). In some cases, the studies are focused on the estimation of the effects of different land-use/cover systems on soil erosion processes (Kinderiene and Karcauskiene, 2016). The soil erosion context in Italy is quite similar to the European framework. Despite of the presence of several publications studying different phenomenologies of soil erosion (Borselli et al., 2008; Della Seta et al., 2009), only few have dealt with the Italian forested areas (among others, Sorriso- Valvo et al., 1995; Porto et al., 2009) or under different land use (Vacca et al., 2000).

There is a limited number of field measurements, throughout the Mediterranean basin, focusing on afforestation in experimental plots (Walsh et al., 1998). Furthermore, many of these field measurements revealed some difficulties related to the extrapolation of soil loss data in areas larger than experimental plots. Hence, alternative approaches, as for example the use of cesium-137 (^{137}Cs) were tested to collect more information and reliable estimates both at plot (Porto and Walling 2012 a, b) and at catchment scale (Porto et al., 2001, 2009).

Concern over accelerated soil erosion and associated land degradation emphasized the need for improved information on rates of soil loss also in terms of selecting management goals and strategies, mitigating forest soil erosion as part of a sustainable forest management plan. Various erosion prediction models and procedures can be used in making these management decisions, such as RUSLE2 approach. Most of this work has been done for standard row-crop agriculture but managing so as to reduce erosion is also a concern for forested situations. It is not clear, however, how well the newer RUSLE2 model with its new cover-management subfactors (USDA-ARS, 2013) can work for these situations. The RUSLE2 (Revised Universal Soil Loss Equation, Version2) model tries to make the estimate as independent of land use as possible (Dabney et al., 2011).

The aim of this work is to evaluate the role of forest stands and their management on mitigating soil erosion processes in a Mediterranean mountainous context in relation to different forest types and treatments, through the application of two different methodologies. In particular, ^{137}Cs tracer technique and RUSLE2 model were used to produce different approaches of estimating soil loss redistribution rates on the same study area selected in Aspromonte National Park, Calabria, Southern Italy. The areas chosen are included in a high beech forest and a pine plantation forest, in order to observe the responses in different contexts.

1.1 The origin of ^{137}Cs radionuclide

Cs is a chemical element belonging to the group '1A' of Mendeleev's periodic table. It is an alkali metal with an only one natural stable isotope, caesium-133. At the time when the atom of the isotope has an excess of nuclear energy it gets unstable and it is called radionuclide. Every atom undergoes the random process of radioactive decay, by which the nucleus loses energy by emitting radiation. The decay rate and thus the half-life can range according to the different radionuclides. Among these, the longest lived are ^{137}Cs with a half-life of 30.1671 years and ^{134}Cs with a half-life of 2.0652 years. ^{137}Cs is considered as the more common fission product by the nuclear fission of uranium-235. It is an artificial radionuclide which emits strong gamma-ray and makes easy the measurements in several environments using devices like gamma detectors, with no need of specific chemical preparation (Ritchie et al., 1974).

As a result of thermonuclear tests, ^{137}Cs was released into the stratosphere where it circulated globally (Owens et al., 1997) and then reached the soil surface (Fig.1).

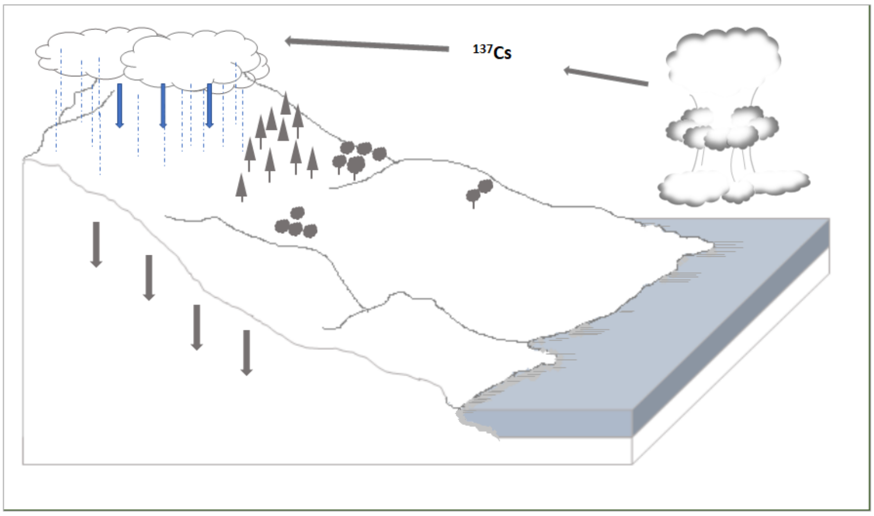


Fig. 1. Example of the ^{137}Cs path in natural environment

A great emphasis was given to its use as a radioactive tracer to estimate soil erosion rates (Porto et al., 2001). In this context, numerous studies have been focused on the spatial distribution and the tracer function of this radionuclide (Ritchie and Mc Henry, 1990; Walling e Quine, 1990, 1992; Golosov et al., 1999; Golosov, 2000; Mabit et al., 2008; Walling, 2012; Giussani and Risica, 2012). Further applications of ^{137}Cs technique were performed by several research groups in England (Walling and Bradley, 1988; Walling and Quine, 1991), Australia (Loughman et al., 1993; Loughran et al., 1988), USA (Ritchie and McHenry, 1973, 1975 and 1990), Canada (Bernard and Laverdière, 1992), providing the basis for the application of the ^{137}Cs approach to measure soil redistribution in natural and agricultural environments. Following a series of refinement and calibration, ^{137}Cs technique was applicated to assess the effectiveness of soil conservation technologies in controlling and mitigating soil erosion and degradation processes (Mabit et al., 2014a).

The ^{137}Cs technique makes use of the global fallout of bomb-derived radiocaesium, which occurred during a timeframe from the mid 1950s to the late 1970s. The temporal pattern fallout of this artificial radionuclide, with a half-life of *ca*. 30.2 years resulting from above-ground thermonuclear weapons testing, was recorded and documented in many regions of the world. After a peak in fallout in 1963, levels have decreased, except for the Chernobyl accident in 1986, minor amounts of fallout have been recorded since mid-1980s.

1.2 ^{137}Cs technique advantages and limitations

The comparing between net erosion measurements on a long term and the estimate derived from ^{137}Cs technique has led to interesting results for soil with forest cover (Porto et al., 2001, 2003, 2004). Many are the reasons why ^{137}Cs is considered a reliable indicator of soil dynamics (Ritchie and Mc Henry, 1990; Mabit et al., 2008):

- The use of ^{137}Cs as a tracer was supported by its capacity to bind tightly and rapidly to the fine fraction of surficial soils and sediments, once reaching the land surface after the fallout. Its concentration was found to be higher in the upper soil horizons as a consequence of the strong deposition relating to the Chernobyl accident (Nimis et al., 1988).
- ^{137}Cs data are useful to measure not only soil erosion but also its spatial distribution within the limits of the studied field;
- Since the vertical distribution of ^{137}Cs in a sediment profile is linked to the known record of fallout in a given region, the deepest occurrence of it in the profile can be equated with the start of ^{137}Cs in the early 1950s, while peaks in activity concentration can be equated with peaks in fallout in 1959 and 1963 (Owens et al., 1997). In fact, its release in the environment goes back to about 55 years ago and given the 30 years half-life the amount in soil are easily detectable. Therefore, long-term trends of soil erosion may be assessed from its measurement (Wicherek and Bernard, 1995);
- This tracer can migrate down the profile over time, but in most cases, it is mainly retained in the top few centimetres of the soil profile (Owens et al., 1996). The erosion rate can be estimated through the measurement of depletion of the ^{137}Cs inventory, relative to the reference, and it will represent the average value for the period from the bomb fallout to the time of measurement (Porto et al., 2001);
- The technique does not require costly and labour-intensive long-term monitoring program but the results can be obtained from a single field-work to collect soil samples to be analysed. Given the minimal area concerning the sampling, no disturbance can interfere with eventual cultivation or silvicultural operations;
- The sampling protocol is relatively simple and it can be completed in a short time, considering the number of soil samples to be collected and the size of the study area.

The application of ^{137}Cs technique however, has some limitations that shall be considered (Mabit et al., 2008):

- The technique is used to make erosion rates estimates on a medium-term (i.e. 50 years) average and thus cannot provide information linked to short-term changes in land-use and management practices;
- In areas located in the southern hemisphere, ^{137}Cs inventories are quite low, hence gamma analyses could require greater count times to obtain a good accuracy for the measurements;
- The routinely used methodology implicates comparison of measured inventories with a reference inventory, representing the inventory associated with a point characterised by neither erosion nor deposition. Given its importance to the accuracy of the estimates of soil redistribution rates obtained, the reference inventory should be accurately established. As such the sampling site used to establish the reference inventory has to provide a representative estimate of the local inventory (Sutherland, 1996);
- The sample representativeness and sampling variability should always be considered; thus, a significant number of samples must be collected (see Owens and Walling, 1996).

Despite the increasing application of this technique, there is still a lack in the research about some feasible relationships between the different types of forest management and the soil loss rate obtained in a forest environment.

1.3 The different estimation models

To convert ^{137}Cs measurements to quantitative estimates of erosion and deposition rates, different approaches can be used (Walling and He, 1999). Each model includes a specific set of parameters despite some of them are common between models. The most commonly used for uncultivated soils is the Diffusion and Migration Model (DMM) which is a one-dimensional transport model (Walling et al., 2002, 2011, 2014) and provides reliable estimates of soil redistribution rates.

For an eroding point, if sheet erosion is assumed, then the erosion rate R may be estimated from the reduction in the ^{137}Cs inventory $A_{ls}(t)$ (Bq m^{-2}) (defined as the ^{137}Cs reference inventory A_{ref} less the measured total ^{137}Cs inventory A_u (Bq m^{-2})) and the ^{137}Cs concentration in the surface soil Cu (t') given by Equation (1) according to:

$$\int_0^t PRC_u(t')\, e^{-\lambda(t-t')}\, dt' = A_{ls}(t) \qquad (1)$$

For a depositional location, the deposition rate R' can be estimated from the ^{137}Cs concentration of deposited sediment C_d (t') and the excess ^{137}Cs inventory $A_{ex}(t)$ (defined as the total measured ^{137}Cs inventory A_u less the local reference inventory A_{ref}) using the following relationship (Walling & He, 1999b)(2):

$$R' = \frac{A_{ex}}{\int_{t_0}^{t} C_d(t')e^{-\lambda(t-t')}dt'} = \frac{A_u - A_{ref}}{\int_{t_0}^{t} C_d(t')e^{-\lambda(t-t')}dt'} \qquad (2)$$

where $C_d(t')$ can be calculated from (3):

$$C_d(t') = \frac{1}{\int_S \; RdS} \int P'PC_u(t')RdS \qquad (3)$$

Given the significant results obtained from the use of the DMM in southern Italy (Porto et al., 2003, 2004), the activity of ^{137}Cs within the soil profile (Walling ar Ie, 1999) was explained in this work through the following equation (4) (Porto et al., 2003):

$$\mathrm{C}(x,t,t') = e^{-\lambda(t-t')} \int_0^\infty \frac{I(t')}{H} e^{\frac{y}{x}} \left\{ e^{\frac{V(x-y)}{2D} - \frac{V^2\,(t-t')}{4D}} \left[e^{-\frac{(x+y)^2}{4D(t-t')}} + \right.\right.$$

$e(x-y)24D$

$(t-t')\times 14\pi D(t-t')-V2DeVxDerfcx+y+V(t-t')4D(t-t')dy$

(4)

where:

$C(x,t,t')$ is the ^{137}Cs activity for the mass depth x and time t', in becquerel per kilogram;

D (kg^2 m^{-4} $year^{-1}$) is the time-averaged estimate of the diffusion coefficient;

V (kg m^{-2} $year^{-1}$) is the downward migration rate;

H (kg m^{-2}) represents the ^{137}Cs initial distribution along the soil profile following the atmospheric fallout;

λ (= 0.023 $year^{-1}$) is the constant of radioactive decay for ^{137}Cs;

x (kg m^{-2}) is the soil mass depth measured from the soil surface downwards;

t (year) is the time since the first fallout;

I(t') in (Bq m^{-2} $year^{-1}$), indicates the ^{137}Cs atmospheric flux at time t';

erfc (u) is the error-function complement (Crank,1975) (5):

$$erfc\,(u) = \frac{2}{\sqrt{\pi}} \int_u^{\infty} e^{-y^2}\, dy \tag{5}$$

Given the continuous input I(t'), the concentration distribution C(x,t) (Bq Kg^{-1}) in the soil profile at time t can be obtained by integrating C(x,t,t') over time t' (Walling and He, 1999) (6):

$$C(x,t) \int_0^t C(x,t,t')dt \tag{6}$$

y is ^{137}Cs concentration measured in the mass of soil (Bq m^{-2});
I (t') is derived from the ratio between ^{137}Cs (x) concentration in the mass depth and the time t;
V is the variation over the time of the downward migration rate (kg m^{-2} $year^{-1}$).
This equation was also significant to the visual representation of the profiles obtained in this study. In Figs. 11 and 12 is reported a peak below the soil surface.

1.4 The USLE/RUSLE1 program

Another way of estimating soil erosion is the use of erosion prediction models to mitigate this phenomenon, as for example the RUSLE2 model. Terranova et al. (2009) and Napoli et al. (2016) compared scenarios of soil erosion by means of GIS at a regional scale for application to the RUSLE model. In a framework in which the availability of input data is not yet mature enough for global scale application, this kind of method, can be fairly accurate to predict soil erosion (Borrelli et al., 2017).

Several applications of this model have been performed for standard row-crop agriculture, but one of the aims of this work is to look into the soil erosion related to forested situations, as for the work that Dissmeyer and Foster (1981) did for the USLE. This conservation planning tool has been demonstrated to do a good job of estimating erosion for many disturbed-land uses and it is still useful for understanding the cause and effect relationship between management practices and erosion. The USLE (Wischmeier and Smith, 1965, 1978) summarized thousands of years of plot research and over 10,000 plot-years of data, collected over seven decades undertaken throughout the United States. It was widely applied for conservation planning purposes on agricultural croplands, based on estimating the average annual soil erosion by water (Renard et al., 2011). The 'Unit Plot' condition was the main concept on which the USLE was based. According to that, the plot was represented by a land parcel 22.1 m in length with a 9% slope, in regular tilled fallow condition and comparable to all the other topographic, cropping, management and conservation practices. Over the years, the several terms used in USLE/RUSLE (cover-management, rainfall, slope steepness, slope length and support practice factors) have evolved to better determine how the conditions of the actual plot could be related to the Unit Plot. The USLE soil loss equation is:

$$A=RKLSCP \tag{7}$$

Where *A* is the computed soil loss per unit area, expressed in the units selected for *K* and for the selected period for *R*; the rainfall and runoff factor *R* is the number of rainfall erosion index units, plus a factor for runoff from snowmelt or applied water where such runoff is significant; the soil erodibility factor *K* is the soil loss rate per rainfall erosion index unit for the considered soil under unit plot conditions; *L* and S are the slope length and steepness factors, related to the unit plot conditions; the cover-management factor C is the ratio of soil loss from an area with specified cover and management to that from an area under unit plot conditions (it ranges from zero in completely non-erodible conditions, to 1.0 in worst-case unit plot conditions); the support practice factor P is the ratio of soil loss with a support practice to that with straight-row farming up and down slope. Several techniques for determining factor values to insert in the USLE equation were reported in the USDA's Agriculture Handbook No. 282 (Wischmeier and Smith, 1965) and No. 537 (Wischmeier and Smith, 1978). Following the release of these handbooks, the USLE became very widely used all over the world. Equally, the planning of further soil erosion research on both natural plots and under simulated rainfall led to improved understanding of the physical processes involved in hillslope sheet and rill erosion.

These several versions of the USLE could be considered "paper" versions, where factors were reported in printed tables and calculation where done by hand (Yoder and Lown, 1995). In the early '80s a program to develop technology to replace USLE was built up, resulting in the computer-based RUSLE model, reported in written form in 1997 (Renard et al., 1997). The first version RUSLE1 was characterised by important improvements in order to allow the application of soil erosion estimation for a wider variety of crops and management practices compared to those in the original USLE database.

The model, defined by a software program operating in DOS-based computer environment, was supported by USDA-ARS through Agriculture Handbook No. 703 (Renard et a., 1997) in which the computer system was described as major conversion of the factor approach described in Agriculture Handbook No 537. The subfactor approach to the calculation of the cover-management factor C in fact, was the main change made in its computer system. This allowed the use of the model for any land use that could be addressed by the USLE subfactors.

To compute soil loss ratio, RUSLE1 uses a subfactor method as a function of five factors:

$$C = PLU \cdot CC \cdot SC \cdot SR \cdot SM \tag{8}$$

where *C* is the overall cover-management factor; the prior land use subfactor *PLU* expresses the influence on soil erosion of subsurface residual effects from previous crops and the effect of previous tillage practices on soil consolidation; the canopy cover subfactor *CC* represents the effectiveness of vegetative canopy in reducing the energy of rainfall striking soil surface; the surface cover subfactor *SC* is the material affecting erosion by reducing the transport capacity of runoff water (Foster 1982), it includes crop residue, rocks, cryptogams, and other nonerodable material that is in direct contact with the soil surface (Simanton et al., Box 1981, Meyer et al., 1971) and it affects erosion by decreasing the surface area susceptible to raindrop impact; the surface roughness subfactor *SR* directly affects soil erosion (Cogo et al., 1984) and indirectly through the impact on residue effectiveness; the soil moisture subfactor *SM* influences infiltration and runoff water. Each subfactor contains cropping and management variables affecting soil erosion. The subfactor approach in RUSLE1 aimed at stop the dependence of the USLE structure on specific land-use data or else calculations would require expensive datasets for each possible combination of land uses (Renard et al., 2011).

1.5 RUSLE1 weak points

The implementation of RUSLE1 program revealed some general weaknesses. First, the model structure was based on science rather than how the user saw things. For example, the soil texture affecting the susceptibility of the soil to develop rills, was thought by the user as a soil property and not as something related to topography, which is where it appeared in RUSLE1 because the tendency to rill controls the response to slope steepness. This example explained the need to approach things more from the user's viewpoint, and not from the modelling viewpoint. Secondly, any user could modify either intentionally or by accident any database value (climates, vegetations, etc), triggering misconceptions such that two users using the same inputs data would get very different results due to the changes made in the database. Finally, the DOS-based interface was already outdated at the time of its delivery, even though the users repeatedly asked for a Windows®-based or similar interfaces (Renard et al., 2011).

1.6 The RUSLE2 program and approach

After several revisions to enhance the interface technology, the model was subsequently delivered as RUSLE2 in 2002 (Foster et al., 2003; USDA ARS, 2008) with its actual diffusion to the field offices in 2004. Based on RUSLE1 characteristics, the new version included improved databases and it was focused on user feedback. This 'hybrid' model is based on the computation of sheet and rill erosion on hillslope through the implementation of empirical equations driven by runoff estimates to determine sediment transport capacity, deposition and sediment enrichment in clay and organic matter (Dabney et al., 2011). RUSLE2 is also used to assess the erosion rates at specific sites, inventory erosion rates over large geographic areas, and estimate sediment production on upland areas that might become sediment yield in watersheds.

In RUSLE2 the conceptual use of USLE factors is retained, basing computations on soil loss estimates linked to 'Unit Plot' conditions and using ratios to adapt predictions to other conditions. Through the improvement of scientific approaches, the calibration of RUSLE2 aimed at reproduce the soil loss ratio for different cropping systems and crop growth stages (Wischmeier and Smith, 1978). This calibration made RUSLE2 erosion estimates for common situations similar to the established values used in the US for conservation compliance assessment.

Furthermore, an important change made in RUSLE2 was the downsizing of the USLE factors (which before the model upgrade were considered to be independent of each other) and the structuring of the information into ‘objects’ or descriptions, applied to both the computer programming and the way the data were entered by the user. For example, all soil-related information is included in a soil description and all vegetation information is included in a vegetation description. The model combines these descriptions with the topographic description to get another description and extracts the information it needs from the descriptions to make erosion computations based on climate information contained in a location description.

The particular aspect is that the databases are maintained at the object level, thus for example a vegetation object contains descriptions of canopy and residue characteristics, growth patterns which RUSLE requires to compute the vegetation’s influence on erosion. It basically takes the information contained in the vegetation description and accounts for its effect on the *L*, *C*, and *P* factors through variables calculated internally by the program, including surface residue cover, surface roughness, canopy cover and soil biomass. The major improvement in RUSLE2 is that the user can enter any number of soil, steepness or management breaks along the slope. In doing so, it considers hillslopes as being composed of three layers: soil, management and topography representing a complex one-dimensional hillslope situation. This ability enabled RUSLE2 to deal with different situation as a management alternative.

The program is currently used by the United States Department of Agriculture-Natural Resources Conservation Service (USDA-NRCS) for conservation planning. It is not clear, however, how well the newer RUSLE2 with its new cover-management subfactors (USDA-ARS, 2013) can work for other disturbed-land or undisturbed situations. The RUSLE2 (Revised Universal Soil Loss Equation, Version2) model makes estimates based on specific site conditions. It allows erosion control practices to be tailored to each specific site and tries to make the estimate as independent of land use as possible (Dabney et al., 2011).

1.7 RUSLE2 database general structure

The database development basically began in 2000, when a USDA-NRCS National Database Manager was given the task of expanding particular sections called descriptions for NRCS use on cropland and pastureland. The expansion included the creation of a single database of soils, climate, vegetations and so on, for each nation.

RUSLE2 model contains databases storing the different data set (climatic data, soil data, etc.,) entered by users and held in the above-mentioned descriptions. For example, a management description is characterised by a list of the field operations and related data including the type of vegetation planted. Given the national scale of the database, it was organized into sections uploaded to the NRCS website for use in local conservation planning (Renard et al., 2011):

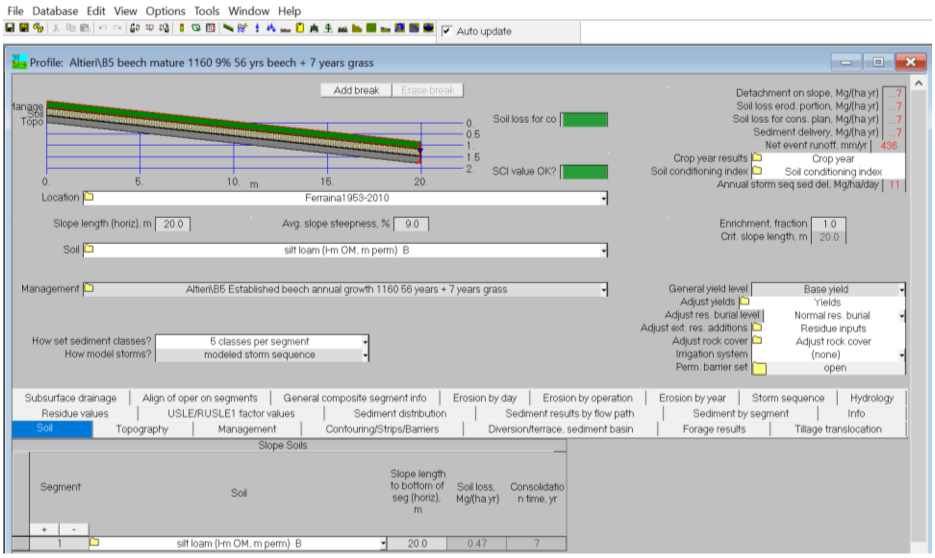
- *Climate records* which stored monthly parameters from a national dataset with calculation of monthly storm erosivity values (EI). The data were organized to provide an average value over a county;

- *Soil records* which stored generic soil descriptions on soil texture downloaded from the NRCS National Soil Information System (NASIS)/Soil Survey Geographic (SURGO) databases available online;

- *Management records* based on the creation of Crop Management Zones (CMZ) namely zones in which climate and other factors thought to control management are assumed to be constant, hence planting and harvest dates are similar in within the same zone.

RUSLE2 interface is characterised by a 'profile view' (Fig. 2) which contains all inputs and outputs related to the erosion calculations for a single situation. The profile includes a single climate, soil and topographic description, along with selection of a single defined management scenario. RUSLE2 also contains the option of a "worksheet view" (not shown), which for a single climate, soil, and topographic descriptions contains a list of likely management options and resulting erosion values. By looking at the worksheet the user can make the comparison of all the management alternatives.

Fig. 2 Example of 'profile view' of RUSLE2 interface.



2 Materials and methods

2.1 Study area

The study focuses on four rectangular plots (15 m × 20 m), materialized in Ferraina site within the Aspromonte National Park, Calabria, Southern Italy (fig. 3). The piece of land Ferraina was purchased by a former Forestry State Company (ex A.S.F.D.) in the year 1951. The total surface area is 897 ha, 38.40 ha of which, including the catchment area of Bonamico and La Verde rivers.

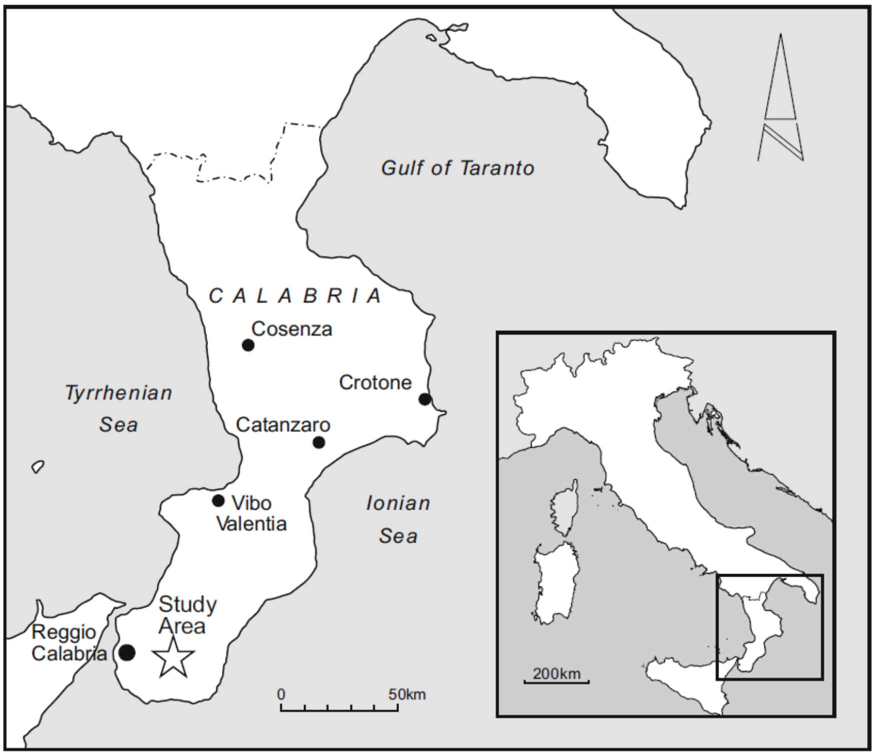


Fig. 3 Location of the study area.

The elevation of the overall study area ranges from 1366 to 1465 m a.s.l. The climate of the area can be classified as 'oceanic temperate' (Rivas-Martinez, 1993) extending up to 1300-1500 m a.s.l, while according to the Köppen's classification (1948) it is classified as Mediterranean (*Csa* – Hot summer Mediterranean climate), with low averages temperatures and dry and hot summers.

In particular as observed from the graph below (Fig 4) the highest temperatures (18,2 °C) correspond to the summer months, namely July and August while the lowest temperatures are recorded in

January, February and December (2,5 – 2,7 – 3,7 °C respectively) (fig.4).

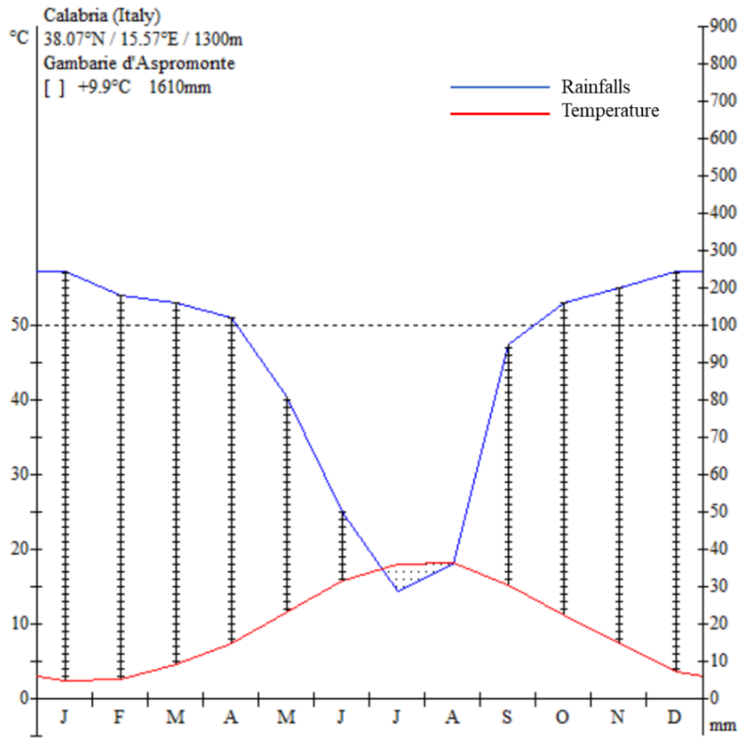


Fig. 4 The 'Bagnouls and Gaussen' Ombrotermic graph of Gambarie meteorological station.

The mean value of annual rainfall, based on the data obtained at Gambarie d'Aspromonte meteorological station (38° 07′ 36″ N, 15° 57′ 08″ E-1300 m a.s.l.) and available for the period 1931-2016, is 1610 mm, with a mean value of annual temperature of 10.6 °C. The precipitation regime in Calabria in strongly influenced by the presence of the Aspromonte massif, together with the coastal mountains whose fronts create particular climate and pressure conditions resulting in seasonal rainstorm events. In the past few decades, because of climate change, heavy precipitation events are increased in frequency and severity (Alpert et al., 2002).

In the last decade, the need of improve the management and preservation activities of soil resource has led the Calabria region as well as the rest of the Italian regions, to examine further geomorphological and pedological aspects of the territory. For this reason, according to the interregional program measure, the soil map has been supplemented with a series of detailed insights as for example the partitioning of soil and subsoil-regions.

The Aspromonte massif is included in the soil region 66.5 (‘Apennine and sicilian ranges on igneus and metamorphic rocks’) and it belongs to the soil sub-region 12 (‘Sila, Serre and Aspromonte mountainous ranges’) (Calabrian soils, 2003). As regard its morphology and lithology it is characterised by crystalline basement rock (Russo et al., 2006), the southern side of the massif presents schist and gneiss, while conglomerates and sub-consolidated sands dominating the western side while granite and granodiorites are distributed on higher elevations (Piovesan, 1994).

The climatic and environmental variability of Calabria region strongly affects the soil types, resulting in a large heterogeneity of soil properties. The main element which unites the soils of this pedo-geographic landscape is the ‘umbric’ diagnostic horizon which is dark surface, soft and rich in organic matter and presents high porosity and a good water hold capacity. The soil type here described, belongs to the ‘Umbrisols’ group of World Reference Base for Soil Resources (1998) including well-structured and shallow soils, moderate to high organic matter content and a coarse texture. The forest stands of the Aspromonte mountains were heavily degraded by the excessive logging activities and the overgrazing that occurred during the post-war years. For this reason, the ex A.S.F.D. planned plantation interventions in most of the land surface funding the operation. Currently, Ferraina site is characterised by ‘fagetum, hot-medium cold subtype’ (Pellicone et al., 2014) spontaneous forest stands, pure or mixed; beech and Calabrian pine are the predominant mixed stands. The hydrography of the forest includes Aposcipo, Ferraina and Butramo torrents leaking into La Verde and Bonamico streams while Menta torrent leaking into Amendolea stream. Over the centuries, Aspromonte forests have been affected by human pressure, that means illegal cuttings to get timber and fuelwood and then to make way for crops. The decrease in woodland took a macroscopic shape from the end of the 18th century to the ‘pre-unification phase’ in Italy (Gangemi, 1985). After the intensive industrial activities in the first half of 1900 and the floods in the ‘50s, which hardly hit the Calabrian lands, a greater awareness of the multifunctionality of the forests spread out.

The other factor contributing to forests and soil degradation was the uncontrolled overgrazing which was the principal means of subsistence for local people.

2.1.1 The experimental plot design

The plots, with slope values ranging from 9 to 22%, were established in 2016. The rectangular shape of each plot (longer axis parallel to the flow direction) aimed at comparing the results of the study with those provided by other models like RUSLE that use the same range of length and slope. The whole area including the plots is covered by forest stands. 'Pine stand 1' and 'Pine stand 2' present the Calabrian pine (Pinus nigra laricio Poir.) species, while 'Beech stand 3' and 'Beech stand 4' present beech (Fagus sylvatica L.) species. Both beech stands had been in long-term mature beech until Beech Stand 3 was cut in 2009 and replanted to grass (Fig. 5).

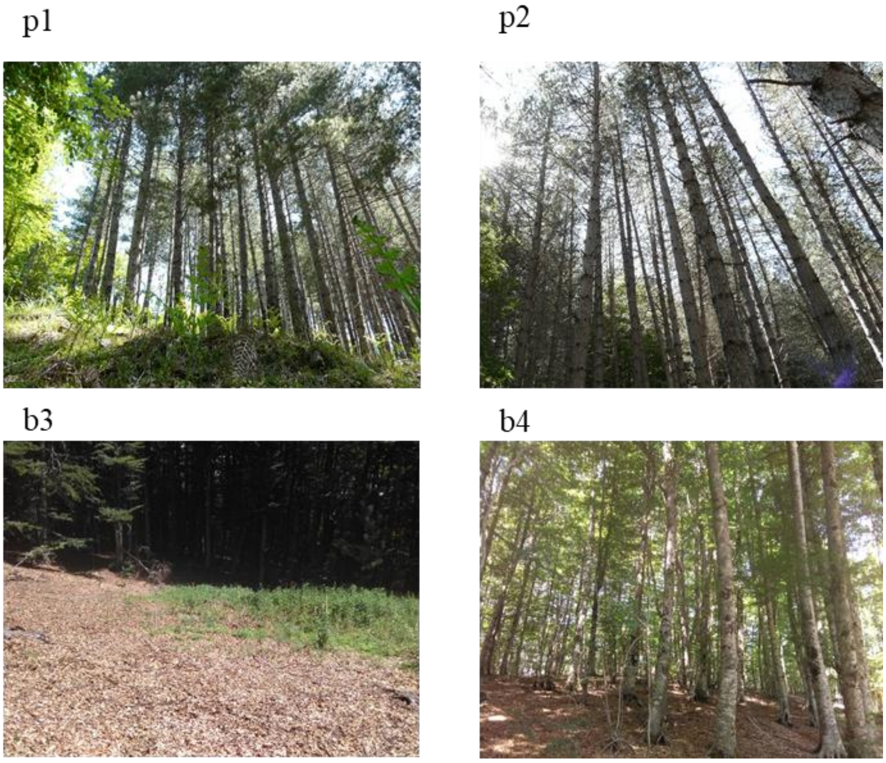


Fig. 5 The pictures show respectively the four study plots Pine stand 1 (p 1), Pine stand 2 (p 2), Beech stand 3 (b 3) and Beech stand 4 (b 4).

To provide more details, the pine stands were previously-neglected poor-quality pasture planted to pine in the late 1960s during a massive re-afforestation program (ca. 150,000 ha) carried out in Calabria. This was one of the first examples of planned intervention made at regional level (Iovino e Menguzzato, 2002a). After the post-war period, the lands set up for forests increased by 32% (Iovino e Menguzzato, 2002a). About a fifth of these lands have been planted with fast-growing exotic broadleaved trees (Eucalyptus x trabutii ed E. occidentalis Villm.) and the remaining part with conifers like Calabrian pine and other Mediterranean pines. The use of conifers was driven by the need to use species able to rapidly provide soil cover and mitigate hillslope erosion, as well as for timber production which was strongly demand on the market (Iovino e Menguzzato, 2002a). Following intense logging activities persisting for ages and numerous fires, at the beginning of 1960s several plantations aimed at soil protection and conservation were carried out in the mountains of the Calabria region.

The beech plots (Beech stand 3 and Beech stand 4) are characterised by seminatural high beech forests (ca. 140 years old) typical of mountainous areas in the Mediterranean basin. In the Aspromonte mountains the beech stands are usually found at elevations between 1000-1950 m a.s.l. The silvicultural treatments applied are those typical of southern Italy beech stands. In the past century the forestry Law (Regno delle due Sicilie, 21.6.1826) included the 'clearcut with reserves' but once the topsoil was regenerated, as reported by Hofmann (1956, 1991), the removal cut was rarely performed. As a cultural heritage of this law, the implementation of the shelterwood cutting treatment was adapted in order to be applied to the beech stands.

For this reason, an intense seed-tree cutting of 50% of standing trees was performed instead of executing a removal cut (Masci et al 1999). As a consequence of floods such as those of 1951 and 1953, a special law to the Calabria region was issued ('Legge speciale per la Calabria, 28 novembre 1955', n.1177). By this law, in the whole Calabrian territory, a series of clearance operations were carried out to gradually reduce the large number of hydrogeological instability episodes. After the last improvement cutting in 1963, no more cutting activities were performed.

If Beech stand 4 was substantially undisturbed with the exception of minimal removal of trees in the past decades, on the contrary Beech stand 3 was clear-cut in 2009 and then left to evolve naturally. In order to collect information on ^{137}Cs fallout on the overall study area, a reference site was also selected within an undisturbed flat area (1366 m a.s.l), located nearby the forested plots (Fig. 6). The site has been in good vegetation for the entire studied period but also in the previous decades.

In details, mesic grasslands are mainly located in several flat areas of the Aspromonte mountains (Spampinato et al., 2008). For this reason, the ideal soil characteristics and the vegetation cover made possible its selection in order to be a representative reference site for both the caesium analysis and the RUSLE2 approach.

Fig. 6 The reference site.

The area which is completely undisturbed since 1950, is basically flat (2% slope) and presents groundcover with grassland vegetation and no tree coverage. Given its proximity to the pine stand plots, the climate and the soil were proved to be the same.

2.1.2 Structural analysis of the study plots

To better represent the forest structure in the plots and to correlate soil erosion estimates provided by ^{137}Cs with the type of forest management in the study area, dendrometric measurements were carried out in the same plots with the exception of Beech stand 3 where no trees are available. All the trees in the plots were georeferenced, and measurements of height and diameter were carried out.

Data were then processed to calculate the values of the basal area (m^2 ha^{-1}), volume (m^3 ha^{-1}), number of plants per plot, and number of plants per hectare. Moreover, in order to represent the vertical and horizontal forest structure within the study plots, measures on tree spatial distribution, canopy insertion height, and tree canopy projection on the forest soil were also carried out.

The Stand Visualization System (SVS) software (USDA, Forest Service) was then used to show the tree spatial distribution for each plot. The SVS software generates images representing the stand conditions and explaining possible silvicultural treatments applied (McGaughey, 1997). The types of data required include plant form characteristics and a tree list describing the species, size and location of each component object.

In addition, hemispherical photos using a fish eye lens were collected and processed with the GLA (Gap Light Analyzer) software (2.0 version, Simon Fraser University, Canada) for calculating the canopy cover and the leaf area index (LAI).

The Leaf Area Index is defined as the projected area of photosynthetic tissue over a unit of land (m^2 m^{-2}) so one unit of LAI is equal to 10 thousand m^2 (Watson, 1947; Chen and Black, 1992).

The photographs are taken skyward from the forest floor with a 180° fish eye lens producing circular images supported by information such as shape, size and location of the gaps in the ground forest. Images are then converted into bitmaps and analysed using the specialized image GLA software. The resulting data are processed to obtain estimates of growing-season light transmission and further measurements such as sun fleck frequency, openness and leaf area (Jonckheere et al., 2004; Frazer et al., 1999). A discrimination threshold was set up to draw a distinction between sky and canopy. The software then will generate a

black and white image representing the foliage and the sky respectively (Fig. 7).

Fig. 7 Example of hemispherical photo vision of the Pine stand 1 generated in black and white to calculate the LAI (to the left) and in colour mode to observe the original version.

Based on the pixel count, the LAI is then calculated. In this work, hemispherical photos were taken in Pine stand 1, Pine stand 2 and Beech stand 4, in 2017. In order to obtain a better LAI estimate, ideal light conditions (cloudiness) and the growing-season phase were chosen to perform the test. The cloudy sky in particular, avoids the direct sunlight and provides for a better contrast between canopy and sky. A total of 18 jpg format photos was taken skyward by placing the camera at points materialized on a 7x7 meters grid. The Gap Light Analyzer software was then applied on the photos to calculate LAI values. In details, two attributes representing canopy cover values were selected: canopy openness percentage and LAI.

2.2 Sampling approach for ^{137}Cs analysis

2.2.1 The sampling protocol

The field survey was undertaken in 2016 and aimed at providing long-term data useful to estimate erosion and deposition rates according to the amount of ^{137}Cs in the soil. A total of nine replicate soil cores was collected for each plot. In details, three transects aligned with the line of steepest slope were established for each plot. In the central transect, three replicate sectioned cores (each one consisting of 14×2=28 layers) were collected in order to obtain information on the ^{137}Cs distribution in soil. Six additional replicate bulk cores were also collected from the remaining two transects (Fig. 8).

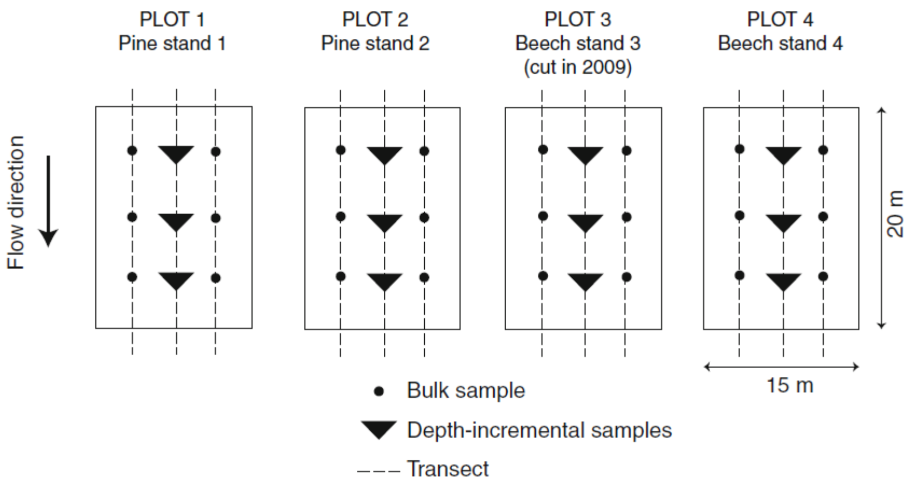


Fig. 8 Experimental plots and sampling locations.

The methodology to collect the samples required the use of a 10-cm-diameter steel core tube introduced to a depth of ca.42 cm. The corer is constructed to cope with stony and compacted soil. To facilitate both the insertion of the core tube and sample extraction, the cutting edge of the tube should have a smaller internal diameter than the tube itself (Walling & Quine, 1993). The tube can be inserted into the soil manually (e.g. using a hammer) or coupled with a mechanical percussion (Figs. 9 and 10) driver in hard soils. In this case the mechanical method was adopted.

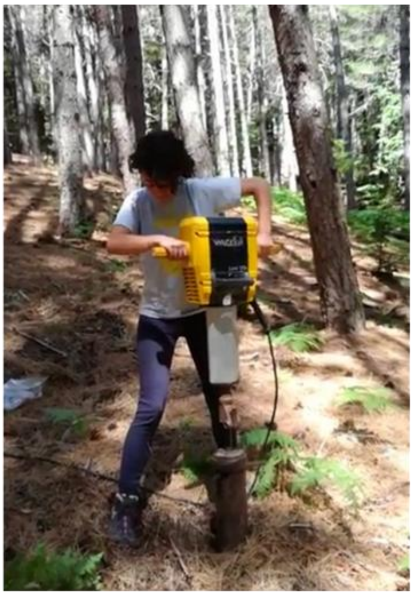


Fig. 9 The motorized percussion hammer

Fig. 10 A steel cylinder core tube used for sampling

The samples were collected in forested areas, in details the sampling was carried out nearby small clearings under the trees to decrease the effect of forest canopy on fallout interception. Such influence has been emphasized in several studies (Bunzl et al., 1995; Strebl et al., 1999; Takenaka et al., 1998). Given that the pine trees on plot 1 and plot 2 were planted at the end of 1960s when most of the ^{137}Cs fallout had already reached the ground, at that time the ground was similar to that of the undisturbed area selected as reference site. As a result, in these two plots, fallout interception was minimal during the period covered by the trees and the ^{137}Cs inventories are then comparable with those obtained in the reference area. In plot 3 and plot 4, possible fallout interception has occurred due to the longer life of the beech stands (ca. 140 years). However, in Mediterranean areas, most of the rainfall is concentrated in winter time when the deciduous species show no foliage cover. Experimental measurements available for deciduous species (see, for example, Lee (1980), Engman (1983) and Bruijnzeel and Wiersum (1987)) documented values of interception around 10-15% for rainfall higher than 15 mm and for forests with uniform canopy cover. These findings, together with the sampling strategy adopted (consisting of taking samples from small gaps), have probably minimized the effect of fallout interception. However, care was also taken to avoid sampling points close to the tree trunks in order to minimize the effect of stemflow on the final ^{137}Cs inventories. In order to obtain information on the ^{137}Cs reference value, a detailed soil sampling campaign was also carried out in the reference area located nearby the study plots (elevation =1400 m a.s.l.).

The reference site is used to estimate the total ^{137}Cs fallout input (reference inventory), which is a key component of the conversion models used to estimate erosion and sedimentation rates from the collected data. In this case, four separate sectioned cores were collected within a large forest gap, using the same procedure as employed for the sectioned cores collected in the study plots. This sampling strategy was adopted to account for the micro-scale variability of the atmospheric fallout (see Porto and Walling (2013)).

The estimate of net erosion from ^{137}Cs measurements is based on the hypothesis that the local fallout of ^{137}Cs was uniformly distributed in space. The ^{137}Cs activities of individual sampling points are compared with that of the reference site to estimate the amount of erosion or deposition that has occurred since initial fallout. It worth noting that ^{137}Cs inventory can usually show some variability (Owens & Walling, 1996) that might be related to some characteristics:

- variability in soil properties that influence ^{137}Cs micro-variability (e.g. infiltration capacity linked to soil bulk density), the effects of topography, vegetation cover and roots, and animal or human disturbance;
- apparent systematic spatial variability at a regional scale, where variations in rainfall and wind flows may have affected the initial ^{137}Cs fallout;

- sampling variability, a function of the surface area over which the samples are collected;
- measurement precision.

To maximize the representativeness of the reference inventory value the following selection criteria should be followed (Mabit et al., 2008):

a) The reference site should not have been disturbed since the beginning of the 1950s. These undisturbed sites should ideally be selected in areas of permanent grassland or abandoned pasture which have not been ploughed for the last 60 years;
b) In order to constitute the initial ^{137}Cs input, the site needs to be selected nearby the study area or at least it should be located as close as possible to the area under investigation;
c) The site should not have been affected by erosion processes, which means a flat area to be selected;

To obtain the spatial distribution of ^{137}Cs as a result of post-fallout soil redistribution, the sampling strategy must integrate the micro-scale spatial variability in ^{137}Cs inventories and the spatial variability of soil redistribution processes (Walling & Quine, 1993). In uncultivated sites, where there is a potential for local variability due to variability in vegetation cover, multiple cores (2 to 3) are commonly collected to represent the individual sample points. The key factor in designing a suitable sampling strategy is to collect representative samples within the field, which reflect the spatial variability of soil redistribution processes.

In this context, the *transect approach* here was adopted. This approach is suitable for fields or small areas with simple topography where no significant across-slope curvature exists and where each point along the transect receives flow from only those points immediately upslope. This method is suited to steep, short and homogeneous slopes. In these situations, a single transect can represent the variability of ^{137}Cs inventories. If significant plan curvature exists, a single transect is not sufficient and a series of equally spaced parallel transects should be used. Transects are generally aligned along the axis of greatest slope, and consist of a sequence of sampling points from the upslope to the down-slope boundary. A distance of 10 to 20 m between the sampling points is generally used. The multiple transect approach, as in this case, is particularly convenient if resources are limited.

2.2.2 Sampling preparation and gamma spectrometry measurements

The protocol used to prepare the samples (Walling & Quine, 1993) includes the following steps:

a) Oven drying of each sample at 105°C for 48 hours;
b) Weight recording of the dried sample;
c) Hand disaggregation. It is important to disaggregate mineral particles, organic concretions and any porous material;
d) Sample sieving through a 2 mm to separate soil particles (<2 mm) from the coarser rock fragments (>2 mm). The bulk density of the soil sample (fine fraction) is accurately calculated for determining the ^{137}Cs areal activity or inventory;
e) Packing of a representative of <2 mm fraction of each samples for analysis. Placement in a 500 g “Marinelli” beaker, or in the case of smaller samplers a plastic dish (65 mm diameter, 15 mm high) or a plastic pot (100 g), in order to measures its activity by gamma spectrometry.

After collection, each core was transported in the laboratory and pretreated for ^{137}Cs measurements. A total of 384 soil samples (336 from the sectioned cores and 48 bulk cores) were collected for the study plots while 120 sample were obtained for the reference area. All the samples were oven dried at 105 °C for 48 h, disaggregated, dry sieved to separate the < 2-mm fraction, and packed in plastic pots or Petri dishes for determination of its 137Cs activity by gamma spectroscopy at the University Mediterranea of Reggio Calabria. The gamma-ray spectrometry equipment used in this study consists of 2

Canberra p-type HPGe detectors, model GX4020. Each detector has 45.6% of the relative efficiency with a resolution of 1.1 keV at 122 keV and 2.0 keV at 1.33 MeV, and it works coupled to a Desktop Spectrum Analyzer DSA-1000 Canberra multichannel analyzer. The spectral analysis is performed using the Canberra Genie 2000 software package. Each detector has been characterized to permit the efficiency calibration by the Canberra's LabSOCS (Laboratory SOurceless Calibration Software) code that performs mathematical efficiency calibrations of Ge detectors, without any use of radioactive sources by the laboratory user. The energy calibration was obtained using a certified multigamma source with a wide energy range (42.8 - 1274.5 keV). Also, a further validation phase was performed using the activity concentration for ^{137}Cs of several standard materials of different geometry to cover the specific range of energy of this radionuclide.

Count times in the detectors were typically approximately 80,000 s, and the ^{137}Cs activities were obtained from the counts at 662 keV. Furthermore, the meaningful results obtained with the use of this model in southern Italy (see Porto et al., 2003, 2014)) strongly support its implementation, which is depicted in Figs. 11 and 12 (see Appendix I listing the rest of the distribution profiles).

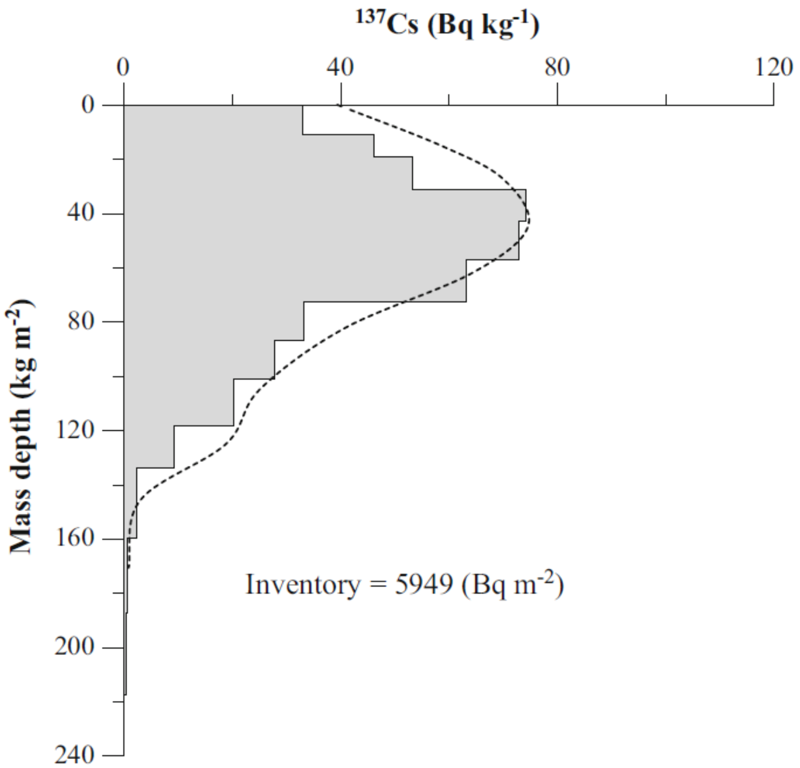


Fig. 11 ^{137}Cs inventory and depth distribution in the reference area.

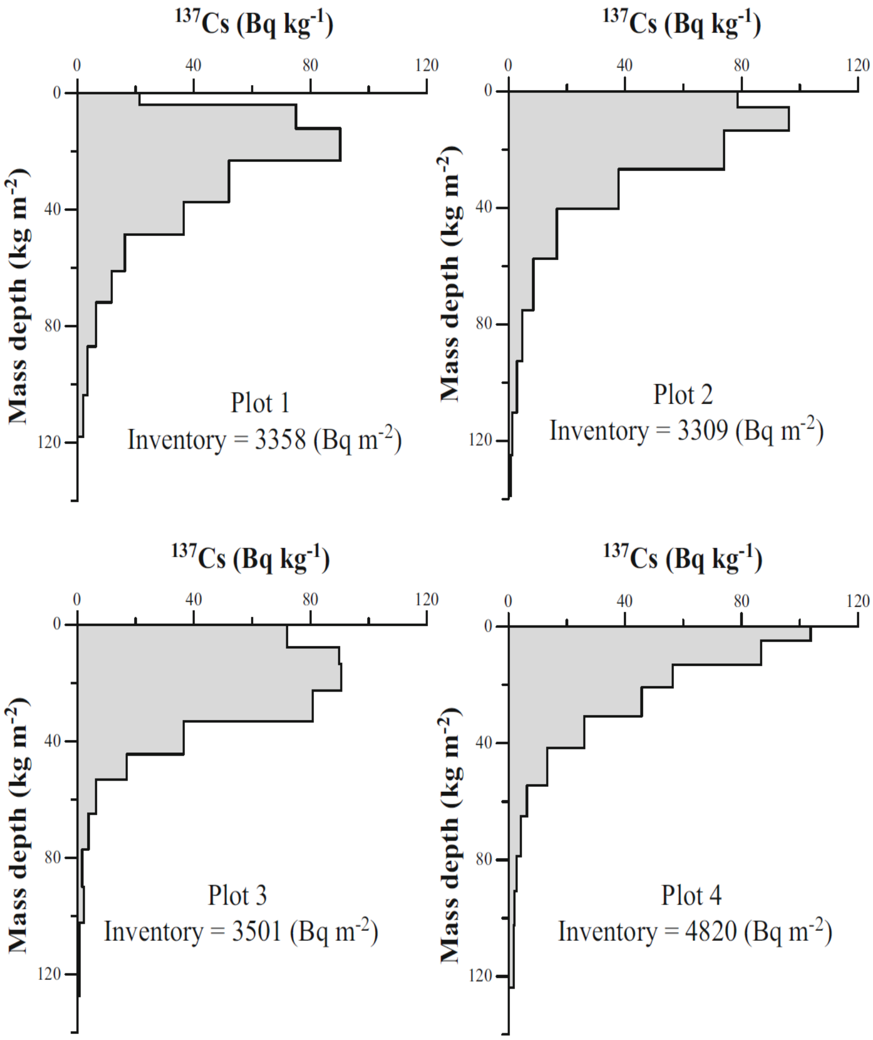


Fig. 12 ^{137}Cs inventory and depth distribution within the study plots.

2.2.3 Particle size analysis procedure

The size distribution of the soil samples was determined using a size analyzer named Laser Particle Sizer *analysette 22* provided by the Department of Agriculture of University 'Mediterranea' of Reggio Calabria. A standard procedure for preparing soil samples for absolute particle size analysis is used (Walling & Collins, 2000):

a) 5 g of soil is weighed into a beaker;

b) 10 ml of distilled water is added to each sample + 5 ml of Hydrogen peroxide; this reaction may need to be controlled by adding a few drops of IMS (Industrial Methylated Spirit);

c) after 2-3 hours, if frothing has ceased, a further 5 ml of Hydrogen peroxide is added and the sample is allowed to stand overnight;

d) the beakers are warmed on a hotplate (starting the process at 80 °C and gradually increasing the temperature to 100 °C) until the reaction is complete and there is a clear supernatant;

e) the whole contents of the beaker are transferred to a centrifuge tube, using a glass rod to clean the sides of the beaker;

f) the tubes are centrifuged at 2500 rpm for 1 hour;

g) the supernatant is discarded and the samples are washed with 2 ml of IMS, before being recentrifuged at 2500 rpm for 30 minutes;

h) the supernatant is discarded and the contents of the centrifuge tube are transferred to a container, using as little distilled water as possible;

i) as soon as the analytical instrument has been set up for use, 10 ml of Sodium hexametaphosphate (6.7 g Sodium hexametaphosphate and 1.3 g Sodium carbonate to 2 liters with distilled water) is added to each sample, which is then subjected to ultrasonic dispersion for ca. 2 minutes;

j) finally, the samples are then analyzed for their grain size composition.

The pre-treatment phase is generally recommended since many soils contain particular aggregates that are not completely dispersed. This is the study case of samples collected from forest soil characterized by large content in organic matter, carbonates and Fe oxides that bind particles together. The following phase requires the use of the Laser Particle Sizer *analysette 22* (Fig. 13) which is an all-purpose measuring device that can be used to determine the particle size distribution of solids in either fluids or gases (suspensions, aerosol) or in drops of liquid (emulsions).

Fig. 13 The Laser Particle Sizer analysette 22.

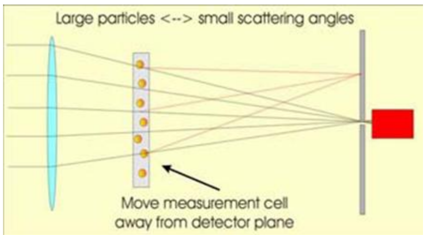
The *analysette 22* is controlled using the MaScontrol software. The exact measurement process is defined using SOPs (Standard Operating Procedures), which permit standardized performance of measurements for frequently recurring sample systems under identical analysis conditions. The analysis of the measurements as well as graphical display of the results can be predefined using templates, allowing standardized reports containing the specific values of interest to be generated as well. MaScontrol is a database-supported program. In other words, all measurements, SOPs, directories and results folders or reports are saved in an SQL database that allows for simple, clear and efficient access to all data.

Laser light encountering particles is deflected from its original direction and scattered at specific angles determined by the size and optical properties of the particles. The scattered light is collected with the help of what is termed a Fourier lens and the angle-dependent intensity distribution of the scattered light is measured by a sensor within the focal plane of the lens. Using the Fourier spectrum obtained in this way, the particle size distribution is calculated in a kind of backward process making use of inversion.

When an expanded parallel laser beam illuminates a collection of particles, a specific pattern is projected onto the focal plane of a convex lens situated further down the beam path. The scattered light pattern is the Fourier transform of the diffracting object and clearly describes the shape and size of the particles. Using an appropriate detector for measuring the pattern, a suitable physical model and the right mathematics for solving the resulting system of equations, one arrives at the particle size distribution.

The convex lens is also called a "Fourier transform lens" and is adapted to a single, defined measurement range. As a result, changing of the measurement range also requires changing of the lens. It applied a known fact of Fourier optics to particle measurement and showed that the Fourier transform of the diffracting object can also be obtained by placing a convex lens in front of the object within a convergent laser beam ("reverse Fourier optics"). Moving the measurement cell (see Fig. 14) causes an effect similar to a zoom function in the Fourier plane. By moving the measurement cell along the laser beam, the measurement range of the *analysette 22* can be optimized and individually adapted to the sample being measured.

a) b)



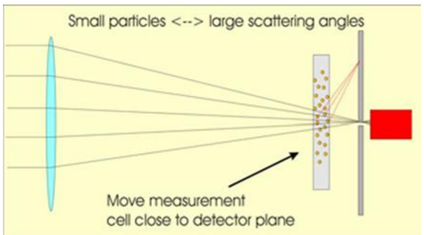


Fig. 14 The position of the sample cell between the Fourier lens and the sensor for the fine (a) and coarse (b) measuring range.

The position of the sample cell between the Fourier lens and the sensor determines the detected particle size range. For the fine measuring range, the measuring cell is located only a few millimeters from the focal point of the Fourier lens, while large measuring cell to detector distances are used for taking measurements in the coarse range.

Particle size analysis (PSA) is performed to measure the size distribution of individual particles generally performed on soil samples to obtain data on the major soil types. Soil particles include a wide size range, varying from stones and rocks down to submicron clays: the size smaller than 2000 µm are divided into three major size groups: sands, silts and clays. According to the aim of the study, there are different systems of classification schemes. The WRB (World Reference Base for soil resources) classification is also used to identify the size groups as follows: clay, silt, very fine sand, fine sand, medium sand, coarse sand and very coarse sand (Table 1).

Table 1 Diameter limits of soil separate (by Foth, 1990).

Name of soil separate	Diameter limits (mm)
Clay	<0.002
Silt	0.002-0.063
Very fine sand	0.063-0.125
Fine sand	0.125-0.20
Medium Coarse sand	0.20-0.63
Coarse sand	0.63-1.25
Very coarse sand	1.25-2.00

The American Society for Testing Materials (ASTM) classification system is used to define soil types for construction purposes. PSA data can be showed in several ways, for example being a cumulative particle size distribution curve, usually exhibiting one or more peaks representing the most prevalent particle sizes (Gee et al., 1986). Hydrologists uses PSA to predict hydraulic properties, particularly for sands. The United States Department of Agriculture (USDA) adopts some defined limits for the basic soil textural classes whose details are given by the Soil Survey Staff (1975).

2.2.5 RUSLE2 model implementation and user inputs

In order to run data in RUSLE2 model, based on the same study plots where the ^{137}Cs technique was applied, the *Climate, Soil, Topography and Management* descriptions were defined in the worksheet, that is:

I) rainfall erosivity description as well as the long-term average monthly temperature and precipitation values, available in the database at the website 'ARPACAL' (Regional Agency for Environmental Protection). The *Climate* description contained both the rainfall erosivity description as well as the long-term average monthly temperature and precipitation values. For purposes of this model, long-term data were usually defined as a 30-year period of record, which was thought to be the longest period over which climatic stationarity can be assumed. These values were extracted for the 1991-2016 period of record from the above-mentioned database. At the beginning, a number of two meteorological stations climate data (Gambarie d'Aspromonte and Roccaforte del Greco) was used to calculate the R-factor. The Analysis of variance of soil erosion values was performed for the above-mentioned stations. As there were no significative differences (p value>0.05) resulting from this analysis, Gambarie meteorological station data were chosen to run the *Climate* description and finally a variation ±5-10-15% of the R factor was considered for the sensitivity analysis.

The erosivity values for the Climate description come out from precipitation data with a higher temporal resolution (at most 30-minute data) that is available from most climate databases. In this case, there was an earlier effort that generated RUSLE2 rainfall erosivity data for the study area, hence the use of those values for the

$$R = \frac{1}{n} \sum_{j=1}^{n} \sum_{k=1}^{mj} (EI_{30})_k \qquad (9)$$

modelling. The R-factor can be defined as the product of kinetic energy of a rainfall event (E) and its maximum 30-min intensity (Brown and Foster, 1987) (9):

R=average annual erosivity (MJ mm ha^{-1} h^{-1} yr^{-1});

n= number of years covered by the data records;

m_j= number of erosive events of a given year j;

EI_{30}= rainfall erosivity index of a single event k (MJ mm $ha^{-1}h^{-1}yr^{-1}$)

k= single event

Average monthly R-factor values were calculated for the 1991-2016 period (Table 2) and then entered in the RUSLE2 climate database.

Table 2 Monthly R factor values for the 1991-2016 period obtained from rainfall data (Mj ha^{-1} mm^{-1} h^{-1}). The last row indicates the average values entered in RUSLE2 climate database.

	MONTHS											
	J	**F**	**M**	**A**	**M**	**J**	**J**	**A**	**S**	**O**	**N**	**D**
YEARS												
1991	156,1	461,7	245,4	179,3	542,0	61,5	132,8	0,0	66,1	250,1	108,7	59,7
1992	47,5	65,5	---	153,4	---	---	---	---	---	---	272,0	507,5
1993	72,5	102,7	390,5	---	---	---	---	---	---	---	---	---
1994	498,8	854,0	0,0	81,8	21,0	20,6	0,0	20,0	117,5	152,8	155,1	74,5
1995	207,3	127,1	192,4	43,6	6,8	0,0	0,0	68,3	173,4	25,3	420,5	372,5
1996	517,3	143,5	257,9	107,7	293,0	0,0	0,0	0,0	441,4	3140,6	79,5	245,6
1997	42,5	26,4	0,0	43,6	18,6	0,0	0,0	463,8	0,0	105,3	17,9	82,6
1998	79,1	134,1	0,0	59,0	75,2	78,8	0,0	103,5	903,0	434,3	251,9	125,3
1999	362,7	42,6	66,4	9,0	---	---	62,8	22,8	234,5	51,8	279,2	189,4
2000	208,6	68,0	24,5	488,1	58,3	0,0	0,0	0,0	827,7	169,7	133,1	130,1
2001	542,5	51,1	31,1	28,2	617,2	80,0	0,0	82,0	0,0	214,4	215,6	227,5
2002	17,5	26,2	28,3	458,7	279,6	250,4	0,0	9,9	603,9	348,5	374,3	328,1
2003	549,6	0,0	0,0	39,3	144,1	60,5	0,0	352,8	108,6	113,4	662,7	5037,1
2004	53,1	0,0	33,0	0,0	0,0	263,1	24,7	0,0	50,0	0,0	141,2	260,4
2005	---	---	---	---	---	---	0,0	14,2	0,0	249,2	8,6	632,0
2006	0,0	0,0	---	0,0	0,0	0,0	0,0	70,8	260,1	---	---	---
2007	---	0,0	19,1	0,0	69,0	20,1	0,0	0,0	72,8	262,8	49,0	143,2
2008	30,0	11,2	188,0	0,0	0,0	336,9	0,0	0,0	636,5	---	819,2	1439,4
2009	1319,2	267,3	252,2	563,6	0,0	471,7	0,0	58,0	366,4	547,0	207,0	225,5
2010	380,7	172,0	431,5	14,4	501,3	121,8	0,0	0,0	---	921,2	739,0	153,1
2011	139,5	343,3	501,3	664,8	31,0	216,2	0,0	0,0	471,1	59,5	1081,6	127,8
2012	134,7	485,1	104,5	198,8	53,2	0,0	370,1	0,0	0,0	131,1	69,2	376,2
2013	870,0	550,1	567,1	267,8	83,1	229,8	0,0	149,2	270,3	0,0	638,2	468,0
2014	556,7	541,0	465,6	542,0	235,3	52,6	179,2	10,1	509,6	181,7	760,7	312,8
2015	910,0	786,2	782,5	733,6	259,2	320,3	398,0	290,5	562,7	615,0	1370,0	1130,1
2016	522,1	296,8	706,7	144,4	487,4	174,5	147,4	351,6	522,1	214,3	391,3	243,5
	342,4	222,2	229,9	200,9	171,6	125,4	54,8	86,1	312,9	372,2	385,2	537,2

II) soil texture to derive 'K soil erodibility' values, that mean the inherent soil properties affecting erosion, runoff and sediment characteristics at point of detachment, that are saved (*Soil*). The best way of deriving these values is from long-term Unit Plots, but this requires substantial cost and effort. As a fall back, if detailed soil texture and organic matter values are known the erodibility can be calculated using the Soil Erodibility Nomograph approach. In RUSLE2 model, rather than either of these, this effort relied on using a soil description based solely on soil texture class, which for all the plots was a silt loam. It should be noted that the maximum range of erodibility values for a silt loam texture is from 0.043 to 0.062, about a median of 0.056 t hr MJ^{-1} mm^{-1}, so the likely error caused by this assumption would be relatively small. To compute soil erodibility, different values of silt, sand and clay fractions are used in RUSLE2 model. Moreover, the software also considers the distribution of the sediment particle classes at the point of detachment and the diameter of the small and large aggregate particle classes.

III) specification of length by modelling plots as single uniform segments and steepness for pine and beech respectively. These data have been collected on each plot during the field surveys (*Topography*). The overall RUSLE2 *Topography* description usually includes dividing a representative hillslope into linear segments starting at the top the slope (where runoff begins) and ending at a concentrated flow channel. For each linear segment the length and the steepness are also specified. In this case, the study was focused on the likely erosion rates near the middles of the relatively small plots. Based on this, the plots were modelled as single uniform segments of 20 m length and steepness of 22% and 10% for Pine stand 1 and Pine stand 2 respectively and 9% for both the beech plots. The LS factor values are obtained considering a portion of constant soil slope s (%) and length (x). To calculate these values, the equation (10) was used:

$$L = \left(\frac{\lambda}{22.1}\right)^m \tag{10}$$

Where *m* takes values based on the slope s:

m=0.5 % to slope ≥ 5%; m=0.4 to 3%<s<5%; m=0.3 to 1%<s<3%; m=0.2% to s<1%

IV) examination of the five RUSLE2 C dimensionless subfactors (2.2) and modelling of the annual addition and decomposition of surface litter as it follows from the annual canopy increase and/or decrease, to evaluate the impact of forests on erosion. The operation and vegetation are set in *Management* description. The operation process describes how it changes cover-managements and soil condition affecting erosion. It is typically represented by common farm and construction activities such as plowing, vehicular or animal traffic, and mowing; it also represents natural processes like germination and growth of volunteer vegetation, as in this case. The vegetation includes temporal values in growth chart, rootmass, canopy cover, fall height and biomass-yield information. In this work, neither of the plots had any specific erosion control practices (e.g. terraces, contour tillage, strip-cropping, etc.), so the P factor is equal to 1.0; no tillage to increase roughness from a naturally relatively smooth value was performed, so the SR (surface roughness) is also equal to 1.0; the PLU (Prior Land Use) subfactor represents rootmass, which will have a relatively large but constant value. The primary RUSLE2 modelling of the impact of forests on erosion is therefore through the growth and death of canopy. As canopy grows, the CC (canopy cover) subfactor will get smaller, reflecting increased interception of rainfall and resulting protection of the underlying soil. As the canopy decreases in the fall season, this effect is reversed, but RUSLE2 models the dying canopy as surface litter, which drives down the SC (surface cover) subfactor. The software then models the decomposition of this litter based on its characteristics and *Climate* temperature and precipitation inputs. Adequately modelling in RUSLE2 the impact of forests on soil erosion is therefore primarily founded on modelling the annual addition and decomposition of surface litter as it follows the annual canopy increase and decrease. This is performed in the *Management* description, and more specifically in the underlying *Vegetation* and *Residue* descriptions (Table 3) wherein the canopy and biomass growth and decrease are modelled, as well as the decomposition characteristics of the resulting surface litter which provided validation of the RUSLE2 modelling efforts.

Table 3 Additional measured values for the pine and beech plots used to assist in the RUSLE2 modelling process.

	PINE		**BEECH**	
	Stand 1	**Stand 2**	**Stand 3 (cut plot)**	**Stand 4**
Total biomass (Mg ha^{-1} yr^{-1})	374	798	1160	1160
Litter half-life (days)	205		205	
Biomass on surface (Mg ha^{-1} yr^{-1})	22		11	
Annual litter addition (Mg ha^{-1} yr^{-1})	9.08		4.54	

The aim of use RUSLE2 in this project, lies in the attempt to assess how does it fit in a forest environment. RUSLE2 in fact, has so far been used exclusively for soil erosion estimates in a typically agricultural range of conditions, for example rangeland, cropland, disturbed forestland, construction sites, mined land and other areas where mineral soil is exposed to raindrop impact and surface overland flow produced by rainfall intensity exceeds infiltration rate (Foster, 2004).

2.2.6 Sensitivity analysis

To make a good comparison between ^{137}Cs and RUSLE2 models, it was necessary to consider a sensitivity analysis in order to see how the results change by changing the parameters of the models. As defined by Saltelli et al, 2004, sensitivity analysis is 'the study of how uncertainty in the output of a model (numerical or otherwise) can be apportioned to different sources of uncertainty in the model input'. Such analysis is mainly reflected by the variation of parameters (Hamby, 1994; Saltelli and Scott, 1997). Sensitivity analysis methods give in general the possibility to identify parameters that do or do not have a significant effect on model simulations for specific cases. These methods are considered an important issue in hydrological and soil erosion models, and furthermore define the impact of errors associated with inputs on model outputs (Blasone et al., 2007). Sensitivity analysis can be used to determine the following points: (1) similarities of a model to the system or processes being studied, (2) the main factors help to provide variability, (3) the relevance of the model components, and (4) the possible interaction between the factors (Saltelli et al., 2000). In practical modelling, the sensitivity analysis is carried out by changing the parameters, the forcing functions, or the submodels and by observing the corresponding response on the selected state variables. Thus, the sensitivity, S, of a parameter, P, is defined as follows (11):

$$S=[\partial x/x]/[\partial P/P] \qquad (11)$$

where x is the state variable under consideration.

The sensitivity analysis on the RUSLE2 model was performed to assess the overall influence of input parameters to the predicted soil loss estimates provided by the model. This was done by taking the current values in the runs as base values, then varying each of the main effects (R value, slope length, slope steepness, management biomass) by ± 5, 10, and 15%, and looking at the change in result caused by each variation.

In the case of RUSLE2 model, the uncertainty is found to be high (Lee and Lee, 2006). These uncertainties can stem from input data (measurement errors, coarse spatial and temporal resolution, missing values), model (parameter, structural, and algorithmic or numerical uncertainty), and stochastic nature of the soil erosion process (Beven and Brazier, 2011; JCGM, 2008). The relative change in the parameter values was chosen based on the certainty of the parameters (Carmona et al., 2017).

A restricted set of model parameters was used for the sensitivity analysis in order to see to what inputs the results were most sensitive. It is often necessary to find the sensitivity at two or more levels of parameters changes as the relationship between a parameter and a state variable is rarely linear.

For each run a 'Base' condition was first set and then one of the main inputs such as R, K, L, S and Biomass were changed by -15%, -10%, -5%, +5%, +10%, +15% while keeping all other inputs at the Base values. For each setting the model was re-run to see how the result (total sediment delivery) changed with the changing input.

3 Results

3.1 Structural data of study plots

Data obtained from the structural surveys are listed in Table 3. Pine stand 1, Pine stand 2, and Beech stand 4 showed, respectively, a mean diameter of 32.31, 33.27, and 36 cm. The same stands showed volume values of 699, 1589, and 1080 m^3 ha^{-1} while the number of trees per hectare was 866, 1500, and 933, respectively (Table 4).

Table 4 Dendrometric values per plot.

Plot	Mean DBH (cm)	Mean H (m)	Basimetric area (m^{-2} ha^{-1})	Volume (m^{-3} ha^{-1})	No. of trees per hectare
Pine stand 1	32.3	19.9	74.1	698.9	867
Pine stand 2	33.3	24.7	138.9	1598.9	1500
Beech stand 4	36.1	22.8	102.6	1080.4	933

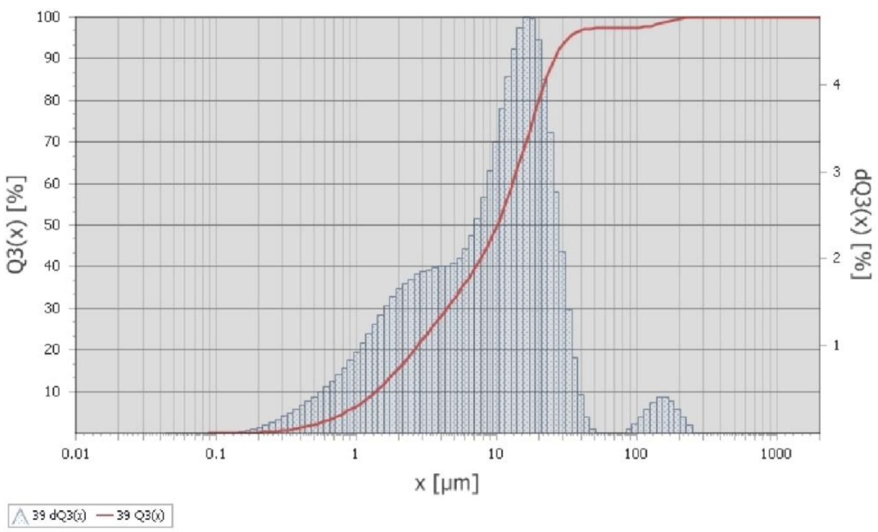
The tree canopy cover determined with the use of hemispherical photos and the GLA software was 52% for Pine stand 1, 63% for Pine stand 2, and 51% for Beech stand 4. The corresponding values of the LAI were 0.79 (m^2 m^{-2}), 1.13 (m^2 m^{-2}), and 0.80 (m^2 m^{-2}), respectively, while the canopy openness produced values equal to 47.83%, 34.98%, 49.37% (Table 5).

Table 5 Canopy cover, canopy openness and LAI values obtained from the GLA software application.

Plot	Canopy cover (%)	LAI (m^2/m^2)	Canopy openness (%)
Pine stand 1	52	0.79	47.83
Pine stand 2	63	1.13	34.98
Becch stand 4	51	0.80	49.37

3.2 Particle size measurements

The particle size analysis was carried out on the first two layers of soil, considering a depth of up 4 cm for each layer (0-2 cm and 2-4 cm depth). A total of 32 laboratory subsamples, two for each soil core previously collected, was prepared by following a standard procedure (see chapter 2.2.3). The measurements where then performed through the use of the Laser Particle Sizer *analysette 22* device and produced a histogram reporting the particle size distribution for each sample. The overall data are set up according to the geometric dimension of the particle and then reported to the x-axis of a coordinate system. The components linked to the size of the individual elements and that represent the shares of each particle within the overall distribution are plotted to the y-axis. Fig 15 shows the result in which the cumulative volume distribution and the probability density distribution for particle size of a subsample are displayed. The fine peak was located around 10-30 μm, whereas the coarse peak shifted around 200-300 μm when coarser distributions were obtained. As all the subsamples belong to soil cores collected in the same study area in which the plots were materialized, the particle size analysis produced histograms revealing similar volumes and probability density distribution of the particle size. For this reason, it was thought to report a single representation



explaining the similar trends.

Fig. 15 Probability density distribution (right y-axis) and cumulative volume distribution (left y-axis) calculated for particle size distribution of subsample 2C (Pine stand 2) belonging to the 0-2cm layer.

The particle size (μm) in logarithmic scale is reported on the x-axis. The *Q* symbol on the left vertical axis represents the proportion of particles; the *dQ* symbol on the right vertical axis represents a relative quantity of particles with the given diameter.

According to the calculations of the particle size analysis, the percentage values of sand, silt and clay fraction for each subsample were derived (Table 6).

Table 6 Example of percentage values of sand, silt and clay for a subsample of each plot.

Plot	Sand (%)	Silt (%)	Clay (%)
Reference 1	30.96	58.95	10.09
Pine stand 1B	27.60	60.33	12.07
Pine stand 2B	17.08	64	18.92
Beech stand 3B	1.39	74.60	24.01
Beech stand 4B	2.43	75.48	22.09

The percentage values were then used to calculate the major soil types of the study plots, through the use of the USDA soil textural triangle (Soil Survey Division Staff, 1993). The corresponding values selected for each subsample, produced a silt-loam texture characterising the whole study area, consisting in well-drained, moderately permeable soils with dark-brown silt loam in the subsurface layer. These soils consist of limestone with sand mixtures and clay deposited by water. The picture below (Fig. 16) shows the location of the values belonging to the subsample of the first soil layer.

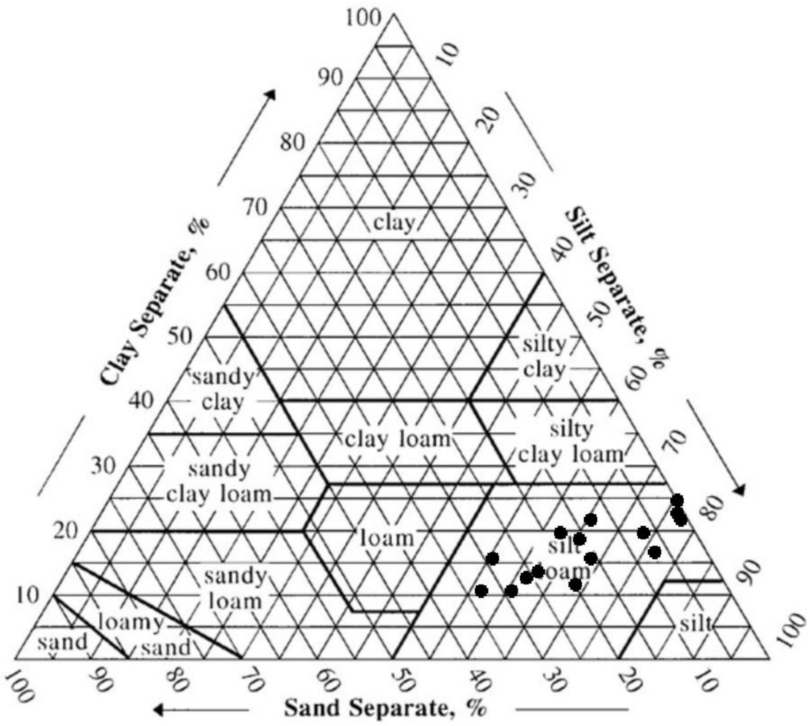


Fig. 16 USDA soil texture triangle of '0-2 cm' subsample group layer. The black spots indicate the silt loam texture extended to all the plots.

3.3 Distribution of ^{137}Cs at study plots and reference site

The four sectioned cores collected within the reference area produced 120 soil samples (2-cm increment). The matching depth increments were bulked prior to gamma assay, and a representative estimate of the local reference inventory and its depth distribution was obtained. This profile, reported in Fig. 10 together with the fitting provided by the DMM, is typical of an undisturbed site (see Porto et al. (2003)). It shows about 90% of the total inventory occurring in the top 15 cm, a peak in activity ca. 6 cm below the soil surface, and a sharp exponential decline in activity below this depth. This profile shows a reference inventory of 5949 Bq m^{-2}, in agreement with what found in other areas of South Italy with similar annual rainfall (see Porto and Walling (2012a)). As mentioned previously, the profile was built on the matching depth increments in order to cover the spatial variability of the atmospheric fallout. However, we considered a 10% uncertainty around this value (see Table 7) to account for the measurements error due to the equipment.

Table 7 ^{137}Cs inventories in the reference area and the study plots.

Inventories	Reference area		Study plots			Net erosion Mean ($Mg\ ha^{-1}\ yr^{-1}$)
Plots	Mean ($Bq\ m^{-2}$)	Range (10% uncertainty) ($Bq\ m^{-2}$)	Mean ($Bq\ m^{-2}$)	Range ($Bq\ m^{-2}$)	Standard deviation ($Bq\ m^{-2}$)	
Pine stand 1			4185	1897-10943	2853	1.16
Pine stand 2	5949	5354-6544	5609	3309-10211	2147	0.26
Beech stand 3			3910	1374-6594	1589	1.34
Beech stand 4			4910	2845-7267	1399	0.68

The values of ^{137}Cs inventory associated with the samples collected from the plots are reported in Table 2 while the ^{137}Cs depth distributions associated with each plot are illustrated in Fig. 11.

The four ^{137}Cs depth distributions in Fig. 11, one for each plot, indicate different inventory values, but they show a similar shape that conforms to that expected for uncultivated soils. Information on the ^{137}Cs inventory values for the study plots, which are provided in Table 1, indicates that 29 out of 36 sampling sites (including the profiles in Fig. 11) show inventory values lower than the reference value pointing out evidence of soil erosion within the plots. Only seven points documented evidence of deposition with ^{137}Cs inventories higher than the reference value. A mean value of 4689 Bq m^{-2} (lower than the reference inventory), obtained from the 36 single points, showed that erosion was the dominant process during the time span (ca. 60 years) covered by the ^{137}Cs measurements.

3.4 RUSLE2 runs

RUSLE2 development of the *Management* and *Vegetation* descriptions and the resulting runs produced four profiles, each one for the same plots related to the ^{137}Cs technique application. The soil loss values given by the single run, related to Pine stand 1, Pine stand 2, Beech stand 3 and Beech stand 4, are reported below (Table 8 a - b). As described above, the dominant erosion-control mechanism in forests is thought to be in the surface cover (SC) subfactor, which will be a function of the production of surface litter and its subsequent decomposition. The *Vegetation* descriptions in the RUSLE2 database are generally for annuals or relatively-short-lived perennials, for which the all or most of the biomass growth each year ends up on the ground as vegetative residue. For trees this is obviously not the case, as much of the annual growth ends up in woody mass. It was thus necessary to develop new *Vegetation* descriptions that provided the correct amount of litter addition each year, based on the measured site litter mass and published data on litter decomposition rates.

These *Vegetation* descriptions were then built into the *Management* descriptions of the long-term tree growth. For Pine Stand 1 and Pine Stand 2 the *Management* descriptions began with existing RUSLE2 descriptions of relatively low-productivity (7 Mg ha^{-1} yr^{-1}) forage, representing the assumed 21 years (1954-1975) of poor-quality pasture followed by planting of the pine plantation and its growth until 2016. The sole difference between the two pine stands is in the modelling of the *Vegetations*, with the Pine Stand 1 *Vegetation* modelled to yield an average surface litter mass equal to the measured 374 m^3 ha^{-1}, while the Pine Stand 2 *Vegetation* matched the measured 798 m^3 ha^{-1}.

The Beech Stand 3 *Management* was modelled as 56 years of beech followed by 7 years of grass, while Beech Stand 4 was modelled as 63 years of beech trees. In both cases the *Vegetations* were designed to yield the measured litter masses, being 1160 m^3 ha^{-1} for Beech Stands 3 and 4.

The Reference site was modelled as long-term grass using an existing RUSLE2 *Vegetation* description. Beech stand 4 is runned like a rotation, thereby is assumed that the profile doesn't change over time. For this reason, no St. Dev. and variability were produced from the model and reported for that plot (see Table 8 b).

Table 8 Estimated results for RUSLE2 climate, soil, vegetation and management inputs in Pine stand plots (a) and Beech stand plots (b).

a)

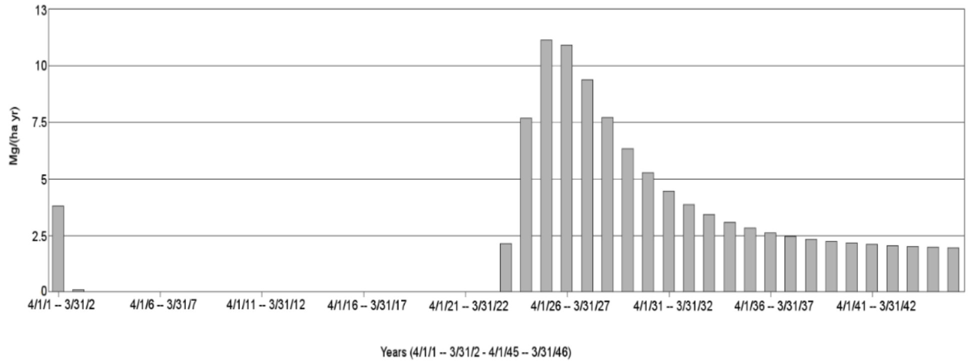
Management			Soil loss (Mg ha^{-1} yr^{-1})		
Plots	**Operations**	**Vegetation**	**Mean**	**St. Dev.**	**Variability %**
Pine stand 1	Begin new growth Basic/general/kill vegetation Begin new growth	New forages/Upper Mid South/Tall Fescue with Bermudagrass. Conifer from planting	2.3	0.45	19
Pine stand 2	Begin new growth Basic/general/kill vegetation Begin new growth	New forages/Upper Mid South/Tall Fescue with Bermudagrass. Conifer from planting	0.24	0.080	33

Management			Soil loss (Mg ha^{-1} yr^{-1})		
Plots	**Operations**	**Vegetation**	**Mean**	**St. Dev.**	**Variability %**
Beech stand 3 (cut in 2009)	Begin new growth	Established forest, beech	0.47	0.19	41
Beech stand 4	Begin new growth	Established forest, beech	0.40		

b)

The picture below (Fig 17 a-b) shows the shape of soil loss trend which is similar in both the stands and follow the growth pattern set up starting from the pre-plantation management. For example, the peak observed in Fig 17 (a) at the year 25th (11 Mg ha^{-1} yr^{-1}) corresponds to the RUSLE2 estimated erosion rates which starts out much higher immediately after planting and then decreases precipitously below the 2.5 Mg ha^{-1} yr^{-1} threshold.

a)



b)

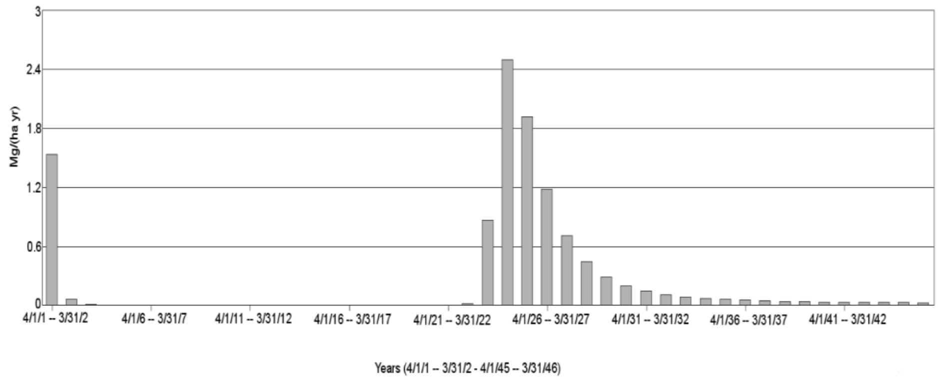
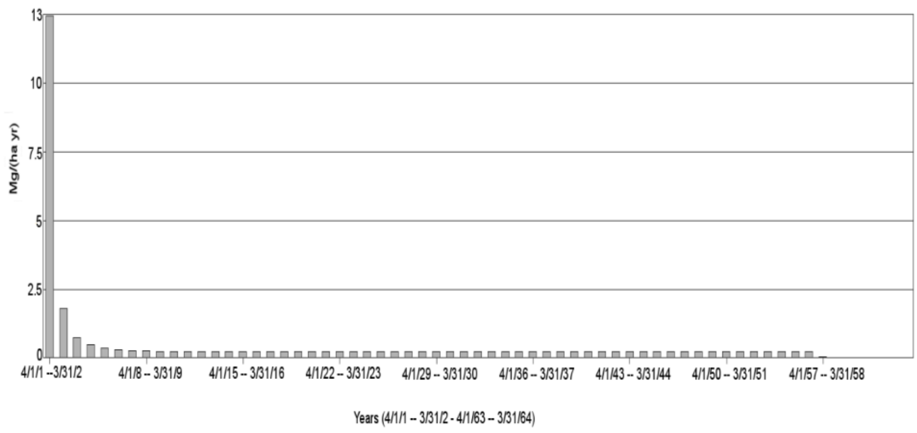


Fig. 17 The soil loss trend estimated in Pine stand 1 (a) and Pine stand



2 (b) profiles.

Fig. 18 The soil loss trend estimated in Beech stand 3 profile (cut in 2009).

3.1.4 Statistical analysis

Based on the results obtained from sensitivity analysis, soil erosion showed to be roughly affected by the variation of R and K, while it was found to be more sensitive to the steepness but not proportionally responding to the variation in steepness. Soil erosion was highly insensitive to the length, whereas it was highly sensitive to biomass parameter (Table 9).

Table 9 Sensitivity of the 5 parameters to the soil erosion in Pine stand 1.

Parameters	Soil erosion (%)
R	1.3
Steepness	2.5
Length	0.01
K	1.1
Biomass	-3.8

An analysis of variance was carried out to make a comparison between the results (Table 10). There are significant differences between values simulated for Pine stand 1 and Beech stand 3 (0.650 and 0.001 respectively) by the two models. Considering the high value of AGB used for parameterising the model in both of these stands, this could suggest that the uncertainty linked to high values of AGB could be high in RUSLE2, confirming what has been stated by Risse et al. 1993; Benkobi et al. 1994, who found out that the cover-management factor (C factor) is the most sensitive parameter in determining soil loss under different agricultural systems and what has been showed by the sensitivity analysis carried out in this work.

Table 10 Statistical comparison (ANOVA) (p>0.05) of soil loss rates obtained from ^{137}Cs technique and RUSLE2 model.

Plots	^{137}Cs	RUSLE2	Sum	Mean	p value
P1	1.16	2.3	3.46	1.73	0.650
P2	0.26	0.24	0.5	0.25	0.001
B3	1.34	0.47	1.81	0.91	0.378
B4	0.68	0.4	1.08	0.54	0.039

The terms P1, P2, B3, B4 indicate the plots studied Pine stand 1, Pine stand 2, Beech stand 3, Beech stand 4.

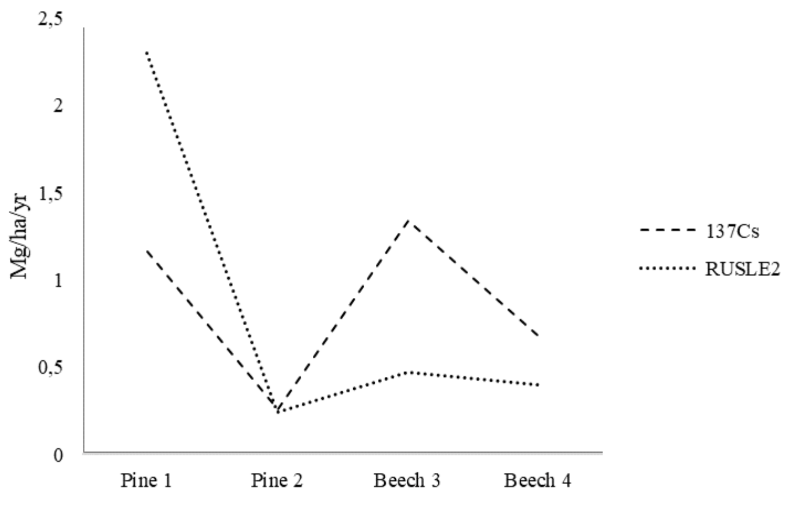


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Fig. 19 Comparison of the soil loss trend produced by ^{137}Cs technique and RUSLE2 model approaches related to the four study areas.

4 Discussion

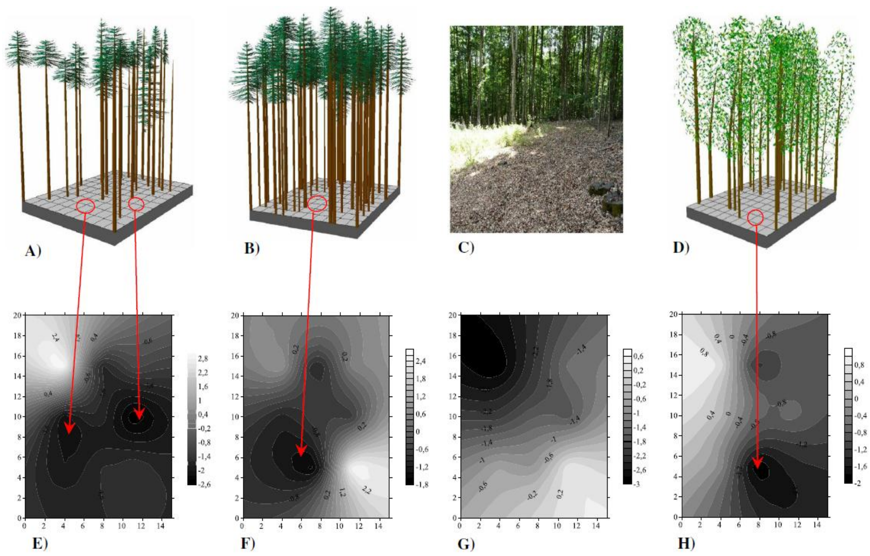
4.1 ^{137}Cs inventories in the study plots

Different values of ^{137}Cs inventories are reported in Table 8. To prior check the significance of these differences, the Friedman test (see Kanji (1993)) was used. This test is a non-parametric test for finding differences in treatments across multiple attempts. The statistical analysis showed that the numbers exceeded the critical value at the 0.05 level of significance, pointing out a significant difference between the ^{137}Cs distributions associated with each plot. An estimate of the average soil redistribution rates for the timeframe extending from the commencement of ^{137}Cs fallout (mid 1950s) to the present was provided by the ^{137}Cs measurements carried out in the nine-sampling point selected in each plot. As described above, the ^{137}Cs technique is based on the comparison between the radionuclide reference value obtained in the reference site and the ^{137}Cs inventories measured in the single points for each study area.

The results of the measurements highlighted that the latter inventories (29 out of 36) were lower than the reference value, indicating that soil erosion was a dominant process during the last ca 60 years. The Table 8 shows a mean value of erosion rates ranging from 0.68 to 1.16 Mg ha^{-1} yr^{-1} which are generally lower than the annual mean values considered as issue of concern by the scientific literature. According to authors like Verheijen et al. (2009) for example, the upper limit of tolerable soil erosion should be of ca. 1.4 Mg ha^{-1} yr^{-1}, even when dust deposition is included in soil formation rates. These findings strongly support the hypothesis that the forests play a key role in mitigate soil erosion and runoff offering protection through the canopy cover and their ability of improving the absorption capacity of soils. These function in fact, are very important to decrease the overland flow and the related water erosion phenomena.

Several differences however, between the four forest stands studied must be observed. The comparison between the four treatments was simplified by the representation of the trees in a spatial visualization, obtained with the SVS software (Fig. 20 a-d) associated to the corresponding spatial distribution of soil erosion rates which is depicted in Fig. 20 e-h.

A first comparison can be made between plot 1 and plot 2 characterised by the same forest species but different biomass and forest cover. In Pine stand 1 having a canopy cover of 52% is observed a mean value of soil erosion equal to 1.16 Mg ha^{-1} yr^{-1} that is higher than that found in Pine stand 2 (0.26 Mg ha^{-1} yr^{-1}) where the canopy cover is 63%. The higher value of soil erosion in Pine stand 1 can be justified by a minor tree density (866 trees ha^{-1}) compared to that measured in Pine stand 2 (1500 trees ha^{-1}) (see Fig. 20 a-b).



Plot1(Pine stand 1) Plot2(Pine stand 2) Plot3(Beech stand 3) Plot 4 (Beech stand 4)

Fig. 20 Spatial representation of stands from each plot (a-d) and corresponding spatial distribution of soil erosion (e-h). (c) shows a photo of the plot 3 where cutting operations were performed (2009).

In a forest stand, the higher tree density in fact, can help to improve the soil protection through the mechanical support function of the root system and a larger surface taken up by the tree canopy intercepting a larger amount of rainfall and reduce the raindrop impact on the ground

(Chapman 1948). Even though both the pine stands belong to the same silvicultural treatment, differences in tree density and forest covers found can probably be due to micro-variability (e.g., stand dynamic evolutions or small disturbances) in that area (Gray and Spies 1996). The forest management applied in the past years has however preserved the protective role of the trees against soil erosion processes. This supports the findings that an increase in forest cover enhance soil protection. The Fig. 20 e-f clearly shows the spots with the highest amounts of soil loss within the plots. This is also supported by the close correspondence with the lower values of canopy cover (see Fig. 20 a-b).

A second comparison can be done between plot 1 and plot 4 having almost the same canopy cover (52 and 51%) but showing different forest species (pine and beech, respectively). Soil erosion in Beech stand 4 is lower (0.68 Mg ha^{-1} yr^{-1}) than that documented in Pine stand 1. This result suggests a better protection against soil erosion offered by the former supported by the explanation related to canopy cover capacity for deciduous species which is greater than for conifers, as stated by other authors (e.g., Helvey and Patric (1965) and Zinke (1967)). Another reason could be found in the root system that is generally superficial and shallow, with large roots spreading out in all directions in the case of the beech, while is relatively deeper and less protective against soil erosion of the first soil layers in the case of the pine. However, the protective role of canopy cover is confirmed by the spatial pattern of soil rates observed in Fig. 20 h. The comparison with the spatial visualization of the trees in Fig. 20 d, shows a clear correspondence between higher erosion rates spots and lower values of canopy cover.

A last comparison can be made between plot 3, originally covered by beech trees (then clear-cut in 2009) and the adjacent beech stand in plot 4. In this case, the plot 3 reports a mean value of erosion rate equal to 1.34 Mg ha^{-1} yr^{-1}, almost twice the beech stand 4. This result is considered as a clear consequence of the clearcutting in terms of soil loss. Moreover, it is clearly expected, because the lack of canopy cover occurred since 2009 may have affected soil conditions and this resulted in a considerable increase of soil loss in these past 7 years. In addition, this short time frame following the clear-cutting has been affected by the occurrence of severe rainfall events which together with the lack of vegetation have further compromised the rate of soil loss in that period.

Although no direct information of soil erosion measurements is available for the recent years, an attempt to model the trend of soil loss using the rainfall erosivity factor (Wischmeier and Smith, 1978) was made. The trend which is observed in Fig. 21 refers to the Gambarie meteo station located near the study site.

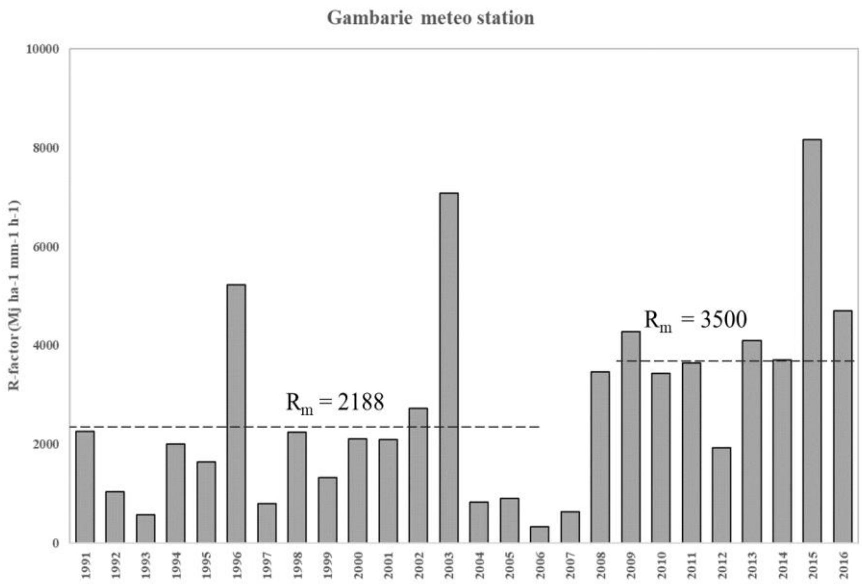


Fig. 21 Rainfall erosivity factor for the Gambarie meteorological station.

The dataset of the 30-min rainfall is available from 1991 to date. The clear-cutting occurred in October 2009, and Fig. 21 indicate that for the set of storm events occurred from that period to the sampling date, the mean value of the R-factor (R_m = 3500 MJ ha^{-1} mm^{-1} h^{-1}) was greater than that related to the period 1991- 2008 (R_m = 2188 MJ ha^{-1} mm^{-1} h^{-1}). The increase is particularly marked for the events that occurred in 2015, prior to the sampling campaign. Similar trends of the R-factor, with a general increase during the last decade, have been also documented by Capra et al. (2017) for other stations located in the same region. The higher values of the R-factor, combined with the tree cutting, showed a general increase of soil erosion even in other areas, emphasizing the importance of vegetation in protecting the soil (see Porto et al. (2006, 2009)). However, further measurements are needed during the next years to establish if the effect of the clear-cutting might be mitigated while the new vegetation will take place.

4.2 RUSLE2 model implementation and comparison with ^{137}Cs technique

The main outcome on RUSLE2 runs show soil loss values for each plot ranging from 2.3 to 0.24 Mg ha^{-1} yr^{-1} as reported in Table 9 a-b. The soil loss values are stored in the 'profiles' worksheet interface (see Fig. 2) created by the software through the combined use of factors required to compute erosion, that are the effects of climate, topography, cover-management, soil erodibility and support practices (Foster, 2004). It is worth noting that the runs were performed for the first time on data collected in forested study areas, which required the use of higher amount of overall live aboveground biomass than the usual entered for agricultural crops (Dabney et., al 2012) in order to adequately reflect the amount of litter produced.

A first comparison can be made between Pine stand 1 and Pine stand 2 profiles (see Table 9 a) having the same management but different aboveground biomass. In Pine stand 1 is observed a mean value of soil loss equal to 2.3 Mg ha^{-1} yr^{-1} that is higher than that found in Pine stand 2 profile (0.24 Mg ha^{-1} yr^{-1}). The higher value of soil loss can be justified by a lower amount of aboveground biomass (AGB) (374 m^3/ha) compared to that entered in Pine stand 2 profile (798 m^3/ha). The aboveground biomass is one of the parameters included in the vegetation description and its variation through time is tracked by the software. Furthermore, the model is particularly sensitive to high levels of biomass amounts, as it was also highlighted by the sensitivity analysis (see Table 10). The AGB value corresponds with the 'Conifer from planting' vegetation description, defining the start of the plantation time. By default, the model considers an initial bare soil condition to start, which explains the high soil erosion value at the very beginning of the simulation for both the pine and beech stands examined (Figs. 17 a and b; Fig. 18). The soil loss trend is then similar for the two stands. Furthermore, the model simulates a very low soil loss during the pre-plantation stage, then a rise in erosion at the age of 22 years old.

This trend reflects the descriptions given to the pine profiles on a 46 years' time span, running from a grassland cover (back to the pre-plantation period) to a tree cover (plantation time span). The process 'Begin growth' identifies the vegetation description 'New forage/Upper Mid South/Tall Fescue with Bermudagrass' which RUSLE starts to use for that time span. Given the origin of the software, the vegetation descriptions stored in the program are built on crop management zones currently used by the NRCS (Natural Resources Conservation Service, USA), for this reason the runs included in this work are partly based on that similar vegetations. The above-mentioned vegetation description corresponds to the pre-plantation timeframe during which the grazing pressure (whose grasslands are sensitive at) is initially low, hence the soil loss almost doesn't change because the grass vegetation increases regrowth. The lack of information on the grassing patterns and its growth cycle makes the pasture modelling very difficult. For this reason, it has been modelled as low-productivity continuous grassland. The 'Basic/general/kill vegetation' process converts the live aboveground biomass namely the grass, to standing residue which is dead plant material. The grass is 'killed' when the trees are planted, hence the 'Begin new growth' process is started with the 'Conifer from planting' vegetation. The absence of soil erosion in those years during which the soil was covered by shrub vegetation, as showed in in figs. 17 a-b, is plausible with the conditions of the areas under observation. Here the solar irradiation is very high and the soil pH is highly acid, therefore the typical shrub vegetation is made of *Rubus ulmifolius* which forms a very dense and tangled layer on the soil surface, particularly inaccessible. The higher soil loss observed in the first years of the plantation can be justified firstly by the disturbance brought by silvicultural operation of plantation which can deeply modify soil conditions. During the plantation activities the shrub vegetation is harvested, the soil is turned over and then seedling put in place. There is, therefore a period during which trees grow and roots develop on a vulnerable bare soil. Moreover, conifer species need a suitable time period to establish themselves and once they are established, their surface cover from the litter takes over. After this first phase, the soil loss gradually decreases in about ten years until it gets almost steady, up to the stand maturity. This is supported by the literature wherein some authors documented that the afforestation using pines is a valid management strategy based on the hypotheses that erosion decrease with the increase of vegetation cover. Elwell et al., (1976) in fact, have experienced that soil erosion is influenced by many variables such as the weeds on surface, organic matter status, tillage and cultivation practices. These variables change the amount of protective vegetation cover over the soil and besides they alter soil erodibility, which is one of the chemical and physical properties of the soil. This needs to be however supported by the protective function provided by the forest stand component.

In fact, although shrub vegetation has protected the soil surface, this was likely to fires. For this reason, in terms of protective function of the soil, shrub vegetation could never be considered the alternative to a forest stand, especially in a Mediterranean environment such as the Aspromonte mountain. However, it can be stated that its occurrence is potentially an important component in reducing soil erosion since the small canopies help to protect the ground from raindrop impact (Abrahams et al, 1995).

In order to decrease soil loss trend, a longer time frame for vegetation establishment and soil protection needs to be observed. For this reason, one point to emphasize could be to focus on a continuous cover forestry, so that the area doesn’t need to be replanted. A continuous cover forestry is in fact, intended as the use of sylvicultural systems whereby the forest canopy is maintained at more levels without thinning or other cut operations (Mason et al., 1999).

A second important comparison can be made between Beech stand 3 (cut in 2009) showing a higher value (0.47 Mg ha^{-1} yr^{-1}) than the Beech stand 4 profile (0.40 Mg ha^{-1} yr^{-1}). The higher value can be justified by the treatment applied in 2009 in Beech stand 3, which caused the reduction of biomass amounts.

This profile is run as a non-rotation, to show the impact of the cutting and the grass growing. The hypothesis made for this situation is to consider how long it takes for the litter production and fine roots near the surface to reach a steady-state after the cutting. The plant litter includes dead plant material (such as twigs, leaves and bark) that has fallen to the ground. The patterns of its deposition are temporally regular and spatially uniform (Berg and Mc Claugherty 2003) as in the case of the beech forest. The fine roots represent a rapidly changing component of forest biomass and given its proximity to the soil surface, they particularly affect runoff and erosion (Foster 2004).

The canopy cover transfers biomass from standing vegetation to plant litter (residue) on the soil surface. Once canopy material falls to the soil surface, RUSLE2 begin to compute its decomposition. Some plants, like corn, lose canopy cover by leaves drooping without falling to the soil surface, which RUSLE also considers. On the contrary, in the case of the beech, canopy grows and falls in the winter and then becomes surface residue and it decomposes based on the temperature and precipitations considered for the study area.

The soil loss observed in Beech stand 3 follows a steady trend (see Fig. 18) for the time frame running from a covering vegetation situation (mature beech stand) to a post-cutting condition. This steady trend could be due to the small size of the studied area: the cut surface in fact, might be irrelevant to the soil loss considering that the plot is surrounded by beech trees providing the necessary soil protection. A good plant cover in fact could be capable of preventing soil erosion (Zhou et al., 2008). Moreover, when the trees are cut the assumption is made that all the roots and surface residue are left in place, and a good-quality grass takes over on top of that. The erosion process appears to be actually linked to the good grassland decrease from what it was in the beech trees, as the annual biomass production for the grass far exceeds that of the trees. Finally, for the last few years of the modelled time the erosion essentially disappears.

In the unmanaged stand (Beech stand 4), wherein the 'Begin growth' process is repeated in one-year cycle, it was assumed that the beech reaches a steady state litter production in a few years, and that it remains constant throughout the remainder of the tree life. RUSLE2 cycles repeatedly through the calculations until the values at the end of the cycle are close to those at the end of the previous cycle, indicating that the equilibrium mass has been reached. Even though for annual crops the total biomass is typically used as a driver of litter production, by contrast in this case for modelling the trees, the linkage between biomass and litter has to be broken, since for the trees the litter production is limited and affected by other factors such as altered soil moisture or different soil surface temperature in gaps (Fetcher et al., 1985; Vitousek and Denslow, 1986), so it stays relatively constant. As the leaves fall to the ground, further material is added to the surface cover fraction which gradually increases. The two approaches were found to be very different in terms of how to use the data collected.

Although the ^{137}Cs soil loss amounts are different from the RUSLE2 model, however analysing the trend among the plots a similar trend is observed.

4.1.2 Model response to sensitivity analysis

Sensitivities to parameters like R factor, Steepness, Length, K factor and biomass were determined through the sensitivity analysis performed on the RUSLE2 model. As reported in tab 9 the values corresponding to the five parameters above mentioned resulted similar for all the plots analyzed. For this reason, in table 9 only the values corresponding to Pine stand 1 are reported. In particular, soil erosion estimates were found to be slightly sensitive to the steepness. This means that the model doesn’t simulate significant variations for the steep slopes because it doesn’t acquire the effect of the increase in steepness on soil erosion process (Liu et al., 2004). An insensitive erosion was observed as regard the length of the slope, that was short enough to have no effect on soil erosion in all the plots. Some studies showed that longer slope could easily trigger solid transport mechanisms (Toy et al., 2002; Soriano, 2013).

The model produces no linear response of the erosion towards the variation of R and K parameters, which are roughly commensurate with the change in values, almost like it is not able to simulate the rainfall effects and the type of soil. This may be due to anomaly of the biomass component. The high sensitivity observed to the biomass parameter confirms that the model is not calibrated for large amounts of biomass. The biomass here considered, belongs to plots materialized in forest stands and therefore includes different vegetal components compared to what is usually considered for the calculations made by the software, namely agricultural crops. In this last case in fact, the components are represented by corn, cotton, grass, fescue and many other perennial vegetation (Dabney and Yoder, 2012; Dabney et al., 2014) whose respective biomass amounts are far more limited than those found in forest lands.

5 Conclusions and future perspectives

In this study two different methodologies, ^{137}Cs technique and RUSLE2 model, used to make soil erosion estimates were applied on data collected in forested plots located in southern Italy, in order to evaluate the responses according to different silvicultural options previously performed. The use of information on the forest structural traits like canopy cover values, tree spatial distribution, silvicultural treatments and dendrometric data has proved to be very important so as to make a correlation with soil erosion estimates obtained from both ^{137}Cs technique and RUSLE2 model.

As regard for the ^{137}Cs measurements, the minimum values of soil loss (0.26 Mg ha yr^{-1}) reported for areas covered by trees (as in the Pine stand 2) underlined the protective function provided by the vegetation against soil erosion.

The higher values of soil loss found in Pine stand 1 and Beech stand 4 (1.16 and 0.68 Mg ha yr^{-1} respectively) where the canopy is lower and irregular, suggest to follow wherever possible, a different management approach when afforestation or silvicultural treatments activities are planned. An example of that, could be to minimize the occurrence of large gaps on the forest ground floor in order to decrease erosion processes. Roose (1988) observed in fact, that the erosion rates are low (0.004-0.05 Mg ha yr^{-1}) in stable forest ecosystems where vegetation provide a uniform protection to the soil, since branches and tree leaves reduce wind and rain energy.

However, this scenario is seen to change in the Beech stand 3, which was clear-cut in 2009. This treatment obviously reduced the effectiveness of canopy, leaving the area uncovered and more prone to erosion. The mean rate documented in this plot was in fact almost double (1.34 Mg ha yr^{-1}) that in the Beech stand 4 where no treatment was performed. Even though the cut activity was carried out in a period during which the rainfall erosivity was the highest in the last 25 years, the test suggests that attention should be paid while planning new silvicultural treatments. Other authors documented similar scenarios (Capra et al., 2017) but further efforts are needed to validate the effectiveness of ^{137}Cs analysis in different forest environments.

One of the attempts in this work was to provide an experimental comparison of ^{137}Cs technique with other methodologies to assess erosion rates, as the use RUSLE2 model. The procedure followed in the RUSLE2 software was tested using data from the same forested research plots formerly selected for ^{137}Cs measurements. The set of descriptions such as climate, soil, topography and management used in this case study produced different trends of soil loss.

In line with the ^{137}Cs results, the low value obtained for Pine stand 2 (0.24 Mg ha yr^{-1}) confirms that a higher cover of vegetation due to a major number of trees found in the surveyed area compared to the Pine stand 1 (2.3 Mg ha yr^{-1}) decrease the erosive events. The erosion estimates for the beech stands resulted in a higher erosion rate (0.47 Mg ha yr^{-1}) in the clear-cut plot (Beech stand 3) compared to the unmanaged stand (Beech stand 4) having a value equal to 0.40 Mg ha yr^{-1}. The lack of canopy cover in Beech stand 3, in fact can result in a major impact of rainfall drops on soil surface which may be significant in a short term (Zuazo and Pleguezuelo, 2009). The assumptions made with the application of RUSLE2 produced a qualitative analysis of the soil loss rates found in the studied plots, by determining whether this model can be possibly applied on various management and ecological conditions.

Obviously, a more elaborate set of descriptions could be created with multiple species and silvicultural treatments to make stronger validations. The purpose of using RUSLE2 is to predict erosion for conservation planning, and its application in a forest environment can help to improve the estimation techniques in this field.

The two methodologies applied in this work were useful to make estimates of soil erosion rates. Although the results obtained were different in the four study plots, low values of soil loss were observed in the areas characterized by a major number of trees and a higher canopy cover percentage (as in the Pine stand 2). These data are in line with many authors who proved that in other forest contexts, the plant cover is a relevant component in prevent soil loss (Francis and Thornes, 1990; Lang 1979).

In parallel, ^{137}Cs technique offered a great potential to provide information on medium-term average of soil loss without the need to disturb the system by installing particular measuring equipment.

The capability of ^{137}Cs as a radioactive tracer to predict soil erosion rates was confirmed and gave important messages to the catchment managers and forestry sector in order to assess how forest stands can improve its protective function curb the frequency of the erosion events.

As a matter of fact, protection is considered one of the most important functions of mountain forests. This function intended as a goal, can be fulfilled through silvicultural interventions aimed at maintaining a stable reference system (Jax et al 1998) and the related forest's structural ability to give protection, hence to preserve the soil. Since forest cover is considered the most effective check of soil erosion process (De Philippis 1970), it is important to draw the attention to those kinds of interventions which enhance the protective function. Among the many, the reforestation could be a valid measure to be applied, especially the Aspromonte terrain which is confirmed to have highly eroded soils. A reasonably increase of vegetation ground cover in these areas would be given right by the canopy as the Cs technique has demonstrated.

However, it should be acknowledged that RUSLE2 model outcomes pointed out a significant role to the shrub vegetation which decreases surface run-off, preserve physio-chemical characteristics and provides stability to the soil (Ciancio et al., 1994). Ground cover like shrubs is in fact strongly correlated to the soil erosion rate, as Cerdà and Doerr (2007) demonstrated in some landscape types with a Mediterranean climate. This last point highlights how important is to examine the effects of floor coverage on soil erosion according to the different management of the forest stands.

This model has showed its critical points right with sensitivity analysis, given that it has never been calibrated for forest vegetation systems. This has resulted in a high sensitivity of biomass parameter to soil erosion. Since this model finds its range of application in cropland systems (Renard et al., 1997; 2011) further trials on several forest study areas could improve the functionality concerning these issues, in order to extend the range of applicability to forested areas.

In conclusion, to date is still not completely possible to use this model to validate soil erosion in forest field. This suggests that further information on management and runs in RUSLE2 are needed to validate the model assumptions as well as additional of ^{137}Cs measurements are required in different forest environments.

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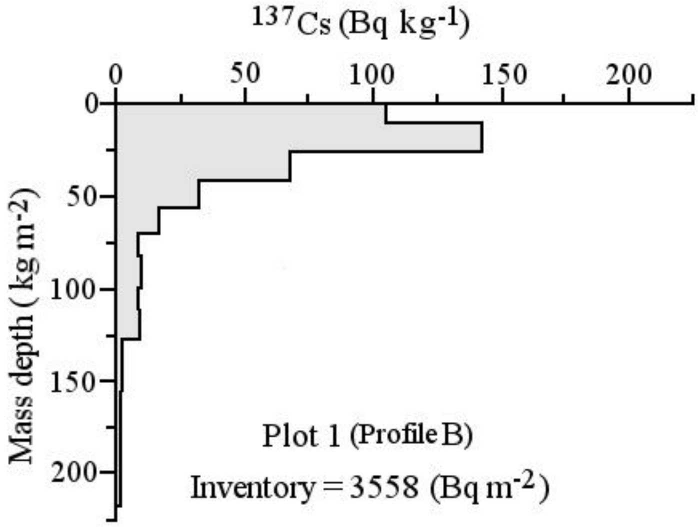
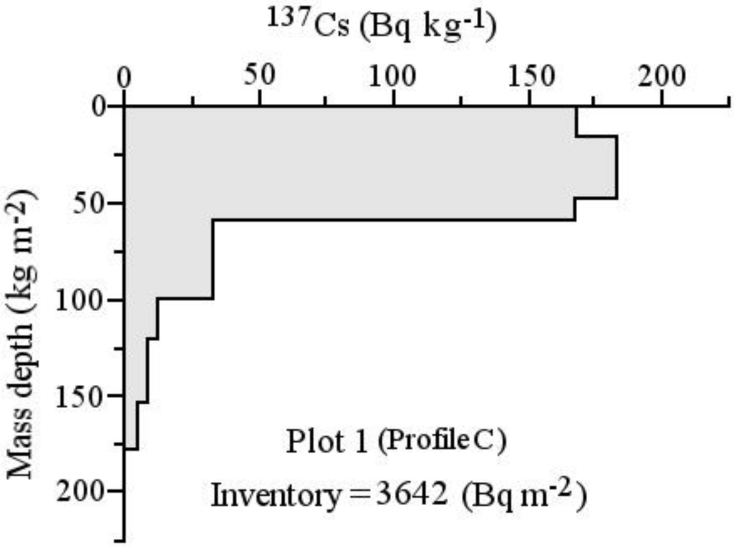
Appendix I

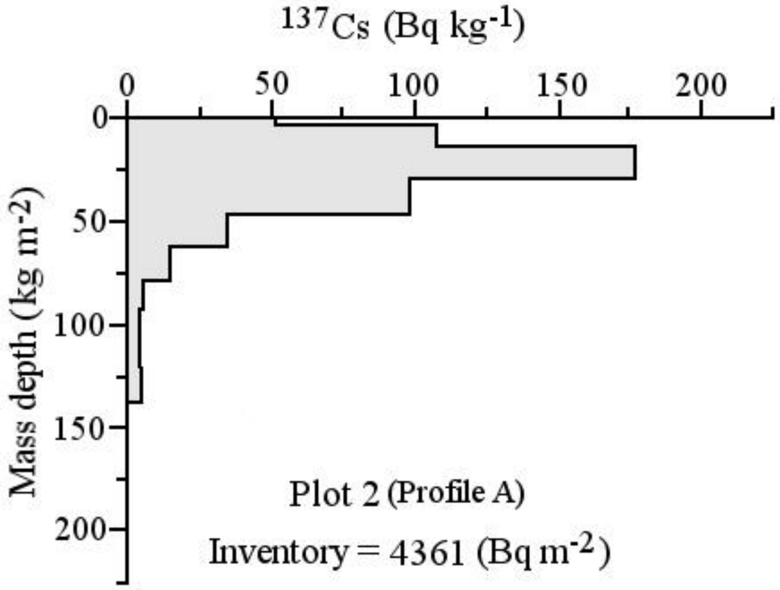
Distribution profiles obtained from the cores belonging to each plot.

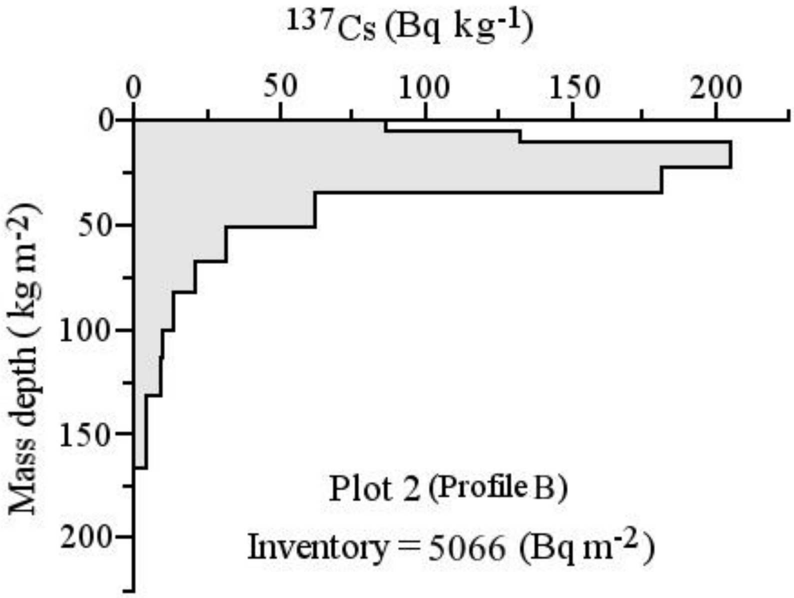


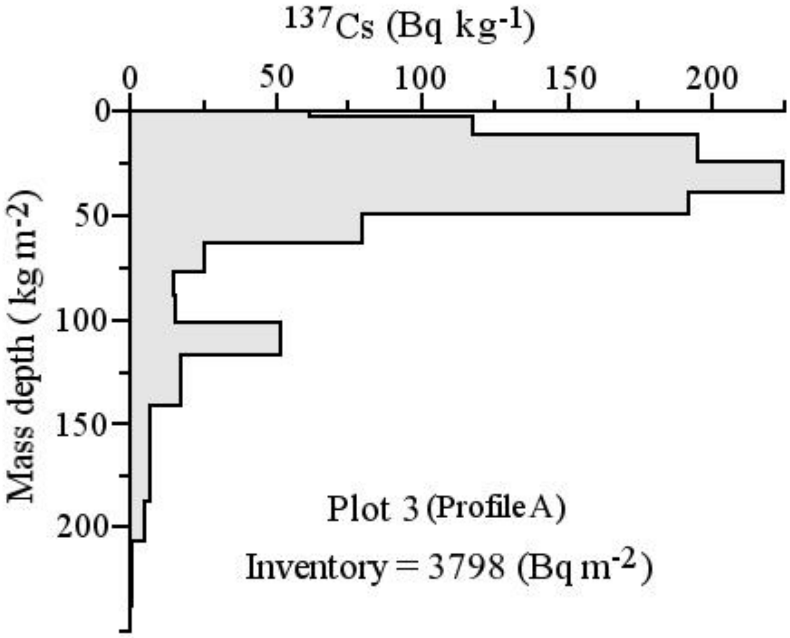
^{137}Cs (Bq kg^{-1})
0 50 100 150 200
Mass depth (kg m^{-2})
0 50 100 150 200
Plot 1 (Profile B)
Inventory = 3558 (Bq m^{-2})



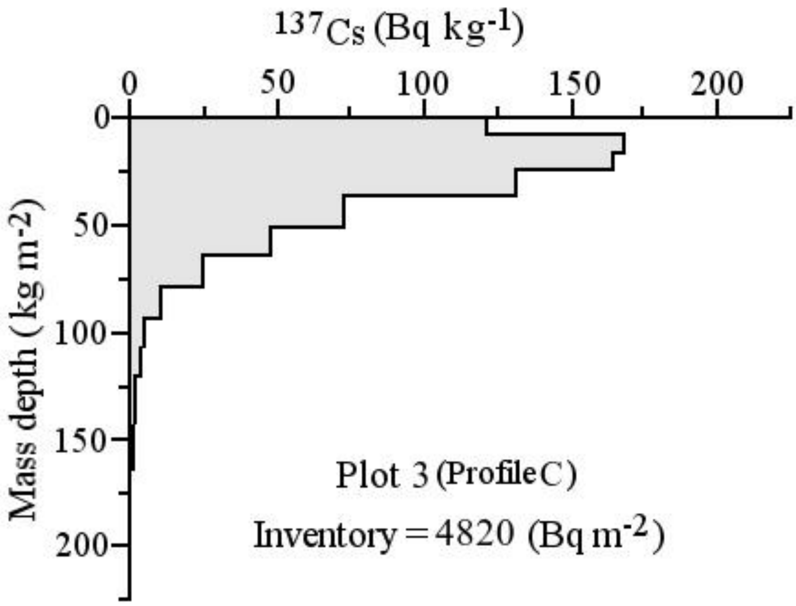
^{137}Cs (Bq kg^{-1})
0 50 100 150 200
Mass depth (kg m^{-2})
0 50 100 150 200
Plot 1 (Profile C)
Inventory = 3642 (Bq m^{-2})


^{137}Cs (Bq kg^{-1})
0
50
100
150
200
Mass depth (kg m^{-2})
Plot 2 (Profile A)
Inventory = 4361 (Bq m^{-2})

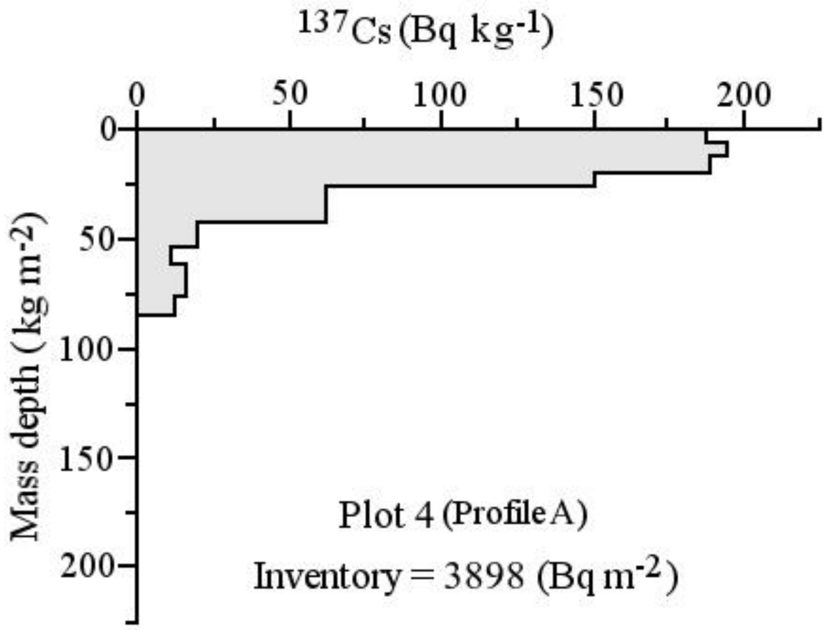

137Cs (Bq kg-1)
0
50
100
150
200
Mass depth (kg m-2)
0
50
100
150
200
Plot 2 (Profile B)
Inventory = 5066 (Bq m-2)

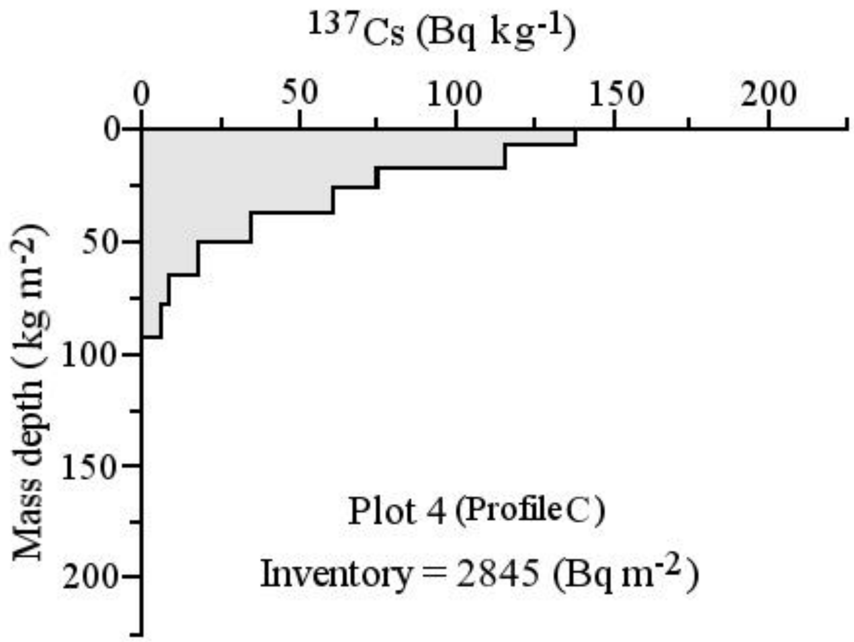


^{137}Cs (Bq kg^{-1})
0
50
100
150
200
0
50
100
150
200
Mass depth (kg m^{-2})
Plot 3 (Profile A)
Inventory = 3798 (Bq m^{-2})



^{137}Cs (Bq kg^{-1})
0
50
100
150
200
0
50
100
150
200
Mass depth (kg m^{-2})
Plot 3 (Profile C)
Inventory = 4820 (Bq m^{-2})


^{137}Cs (Bq kg^{-1})
0 50 100 150 200
Mass depth (kg m^{-2})
0 50 100 150 200
Plot 4 (Profile A)
Inventory = 3898 (Bq m^{-2})


^{137}Cs (Bq kg^{-1})
0 50 100 150 200
Mass depth (kg m^{-2})
0 50 100 150 200
Plot 4 (Profile C)
Inventory = 2845 (Bq m^{-2})

Appendix II

Particle size measurements results of soil subsamples (layer 0-2 cm) generated from the Laser Particle Sizer *analysette 22* device.

Reference 1

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ANALYSETTE 22 MicroTec plus

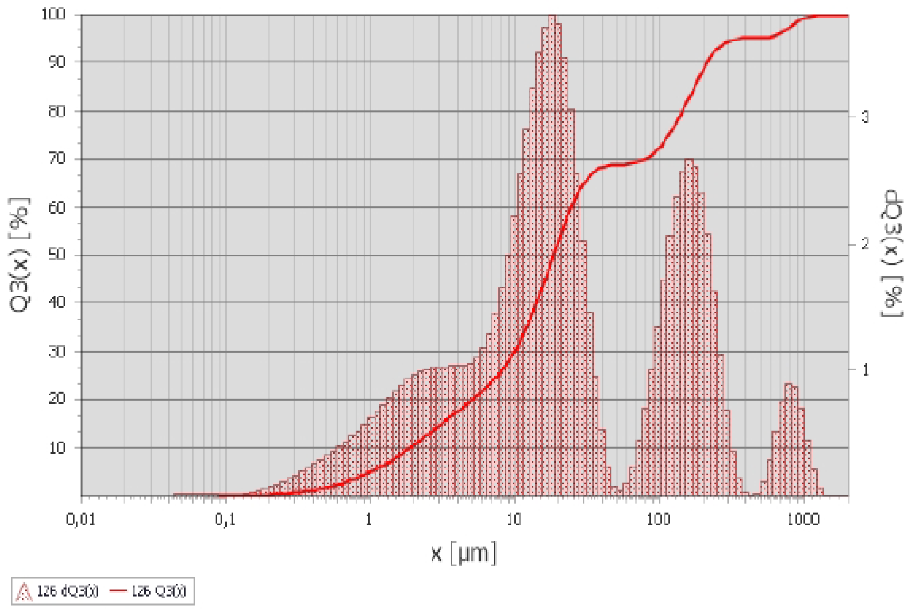
FRITSCH

Meas.No.	126	**Date**	giovedì, 15 dicembre 2016 12:52
Attr. 1		**Attr. 2**	
Attr. 3		**Attr. 4**	

Calculation	Fraunhofer	**TradeOff**	broad (1000,0)
Refractive Index	---	**Absorption Index**	---

Scans Fine 100 **Scans Coarse** 100 **Channels** 102 **Beam Obscuration** 13,0 %

Meas. Range 0,08 [µm] - 2000,00 [µm] **Pump** 60 % **Ultrasonics** 100 %



Mode 18,49 µm **Span** (d90-d10)/d50 11,41 **D[4,3]** 91,8 µm

Q3(x) [%]	x [µm]
5	1
10	**1,9**
25	7,6
50	**18,5**
75	116,3
90	**213,9**
95	334,5
99	968,1

Reference 2

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FRITSCH

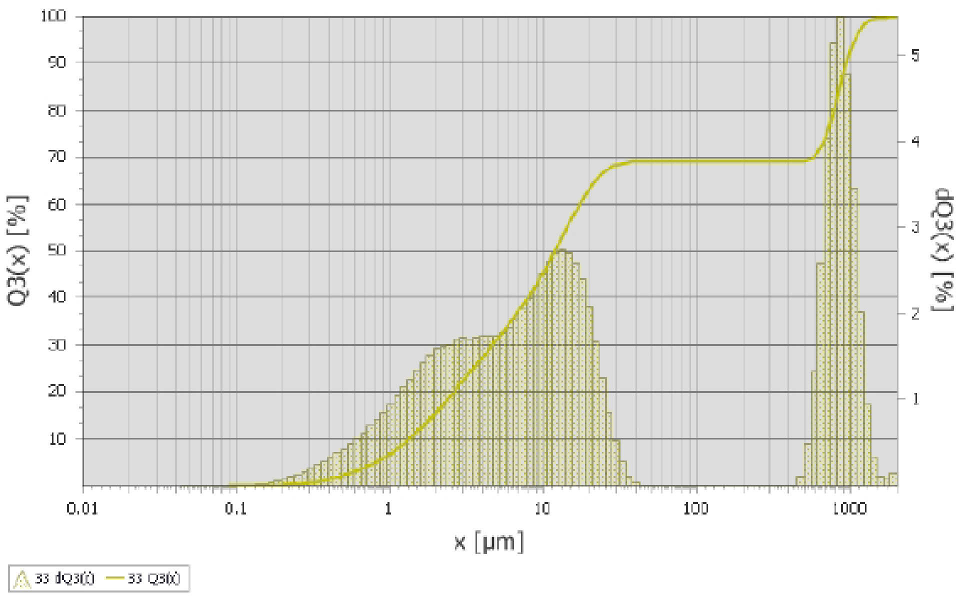
ANALYSETTE 22 MicroTec plus

Meas.No.	33	**Date**	mercoledì, 26 ottobre 2016 02:35
Attr. 1		**Attr. 2**	
Attr. 3		**Attr. 4**	

Calculation	Fraunhofer	**TradeOff**	broad (1000.0)
Refractive Index	---	**Absorption Index**	---

Scans Fine 100 **Scans Coarse** 100 **Channels** 102 **Beam Obscuration** 10.0 %

Meas. Range 0.08 [µm] - 2000.00 [µm] **Pump** 60 % **Ultrasonics** 100 %



Mode 854.25 µm **Span** (d90-d10)/d50 79.25 **D[4,3]** 278.4 µm

Q3(x) [%]	x [µm]
5	0.8
10	**1.3**
25	3.5
50	**11.9**
75	703.5
90	**949.4**
95	1°071
99	1°313.1

Reference 3

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FRITSCH

ANALYSETTE 22 MicroTec plus

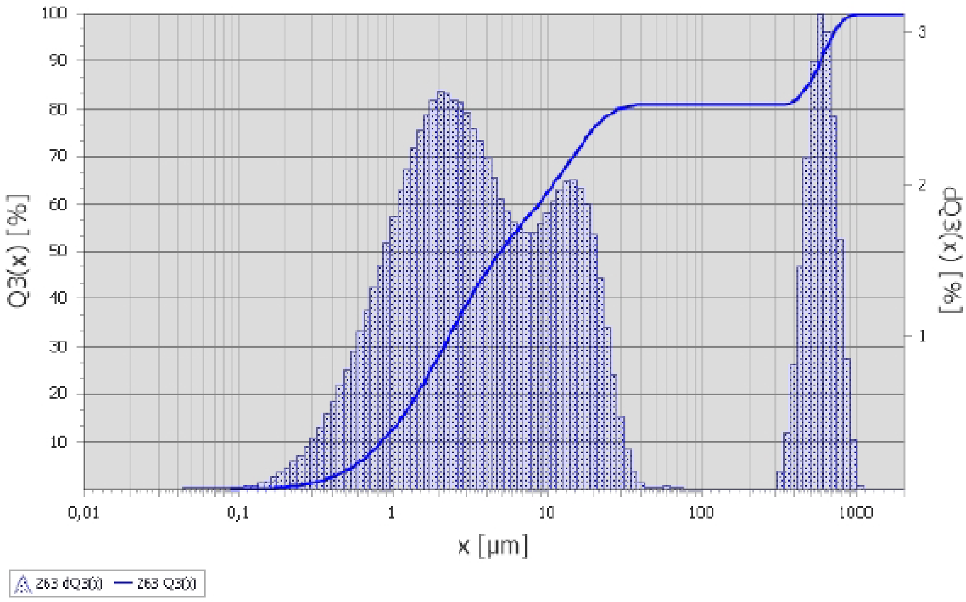
Meas.No.	263	**Date**	giovedì, 26 gennaio 2017 10:05
Attr. 1		**Attr. 2**	
Attr. 3		**Attr. 4**	

Calculation Fraunhofer **TradeOff** broad (1000,0)

Refractive Index --- **Absorption Index** ---

Scans Fine 100 **Scans Coarse** 100 **Channels** 102 **Beam Obscuration** 12,0 %

Meas. Range 0,08 [µm] - 2000,00 [µm] **Pump** 60 % **Ultrasonics** 100 %



Mode 597,61 µm **Span** (d90-d10)/d50 117,72 **D[4,3]** 120,0 µm

Q3(x) [%]	x [µm]
5	0,6
10	**0,8**
25	1,8
50	**4,9**
75	18,9
90	**581**
95	685
99	844,9

Reference 4

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FRITSCH

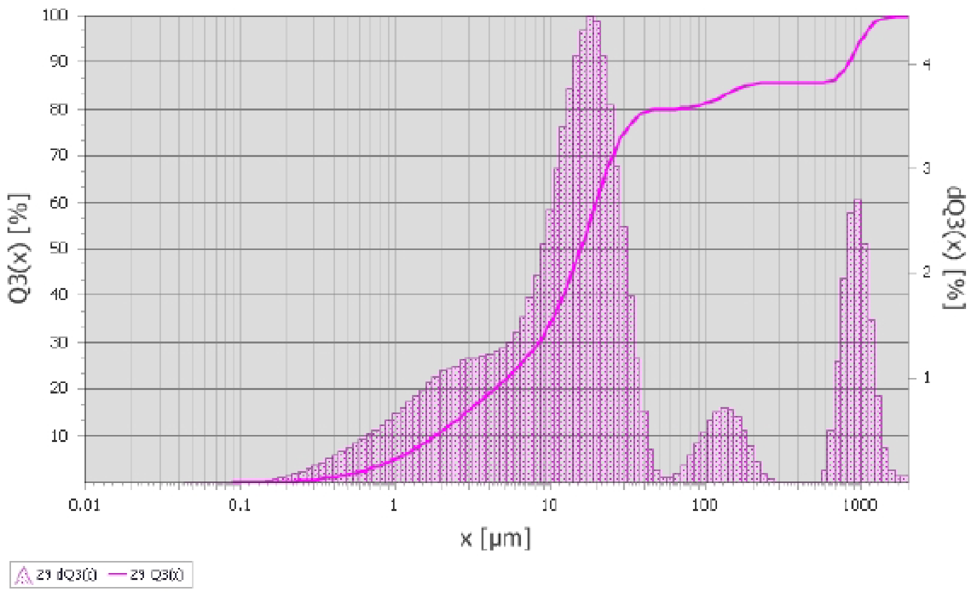
ANALYSETTE 22 MicroTec plus

Meas.No.	29	**Date**	mercoledì, 26 ottobre 2016 01:18
Attr. 1		**Attr. 2**	
Attr. 3		**Attr. 4**	

Calculation	Fraunhofer	**TradeOff**	broad (1000.0)
Refractive Index	---	**Absorption Index**	---

Scans Fine 100 **Scans Coarse** 100 **Channels** 102 **Beam Obscuration** 11.0 %

Meas. Range 0.08 [µm] - 2000.00 [µm] **Pump** 60 % **Ultrasonics** 100 %



Mode 18.49 µm **Span** (d90-d10)/d50 53.59 **D[4,3]** 155.0 µm

Q3(x) [%]	x [µm]
5	1
10	**1.8**
25	6.3
50	**15.7**
75	30
90	**841.8**
95	1°017.2
99	1°289.5

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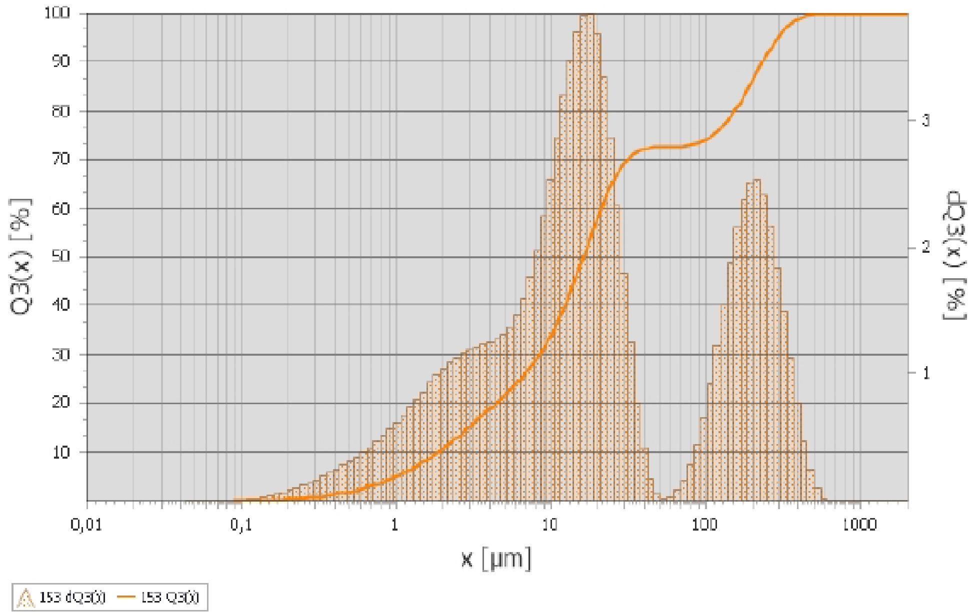
ANALYSETTE 22 MicroTec plus FRITSCH

Meas.No.	153	**Date**	venerdì, 16 dicembre 2016 12:26
Attr. 1		**Attr. 2**	
Attr. 3		**Attr. 4**	

Calculation	Fraunhofer	**TradeOff**	broad (1000,0)
Refractive Index	---	**Absorption Index**	---

Scans Fine 100 **Scans Coarse** 100 **Channels** 102 **Beam Obscuration** 12,0 %

Meas. Range 0,08 [µm] - 2000,00 [µm] **Pump** 60 % **Ultrasonics** 100 %



Mode 17,33 µm **Span** (d90-d10)/d50 14,22 **D[4,3]** 67,7 µm

Q3(x) [%]	x [µm]
5	1
10	**1,9**
25	6,3
50	**16,2**
75	111
90	**232,8**
95	293
99	402,6

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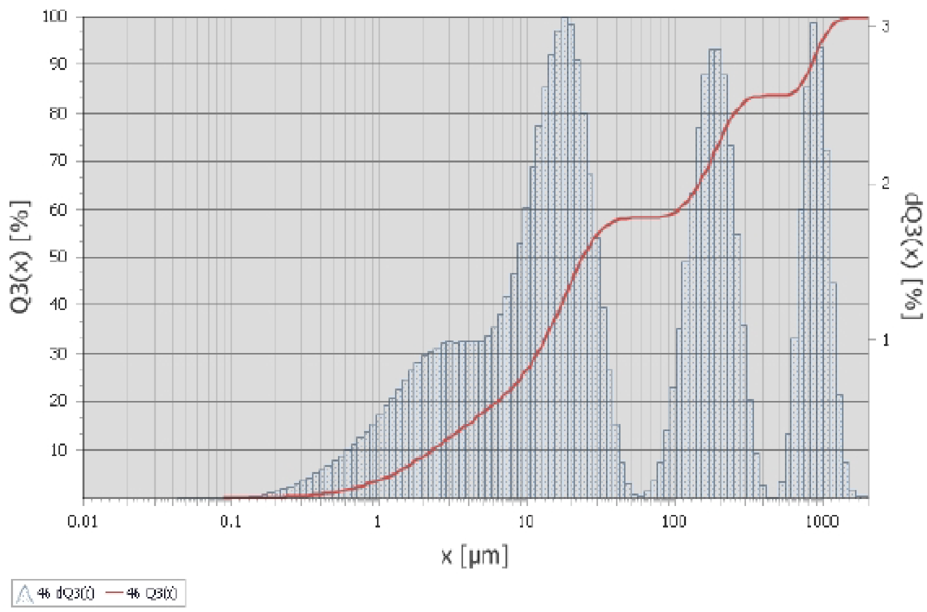
ANALYSETTE 22 MicroTec plus

FRITSCH

Meas.No.	46	**Date**	mercoledì, 26 ottobre 2016 03:52
Attr. 1		**Attr. 2**	
Attr. 3		**Attr. 4**	

Calculation	Fraunhofer			**TradeOff**	broad (1000.0)		
Refractive Index	---			**Absorption Index**	---		
Scans Fine	100	**Scans Coarse**	100	**Channels**	102	**Beam Obscuration**	12.0 %
Meas. Range	0.08 [µm]	-	2000.00 [µm]	**Pump**	60 %	**Ultrasonics**	100 %



Mode 18.49 µm **Span** (d90-d10)/d50 35.08 **D[4,3]** 203.2 µm

Q3(x) [%]	x [µm]
5	1.2
10	**2.3**
25	9.1
50	**23.9**
75	205.9
90	**837.3**
95	991.3
99	1°223.8

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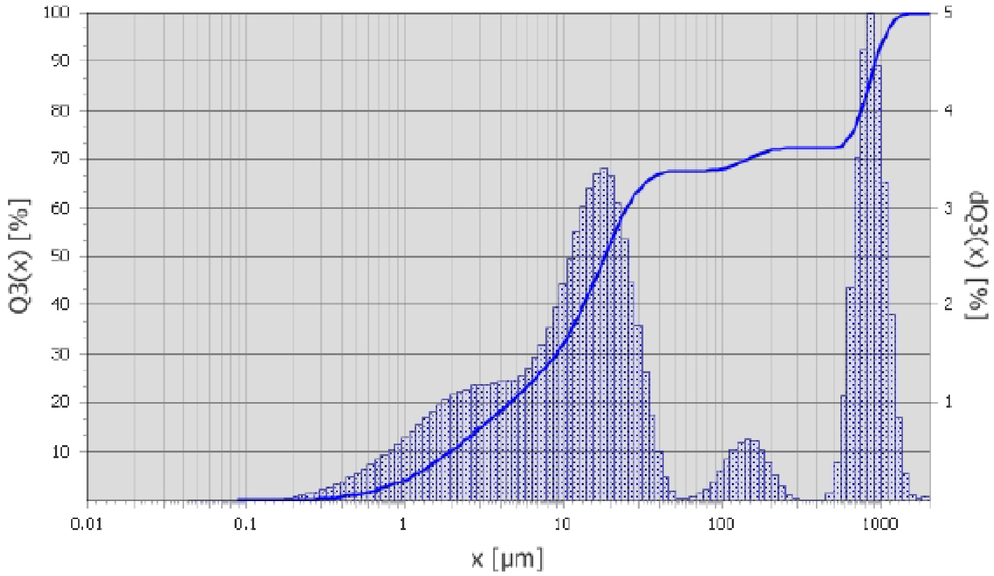
FRITSCH

ANALYSETTE 22 MicroTec plus

Meas.No.	26	**Date**	mercoledì, 26 ottobre 2016 01:01
Attr. 1		**Attr. 2**	
Attr. 3		**Attr. 4**	

Calculation	Fraunhofer		**TradeOff**	broad (1000.0)	
Refractive Index	---		**Absorption Index**	---	
Scans Fine	100	**Scans Coarse** 100	**Channels** 102	**Beam Obscuration**	11.0 %
Meas. Range	0.08 [µm] - 2000.00 [µm]		**Pump** 60 %	**Ultrasonics**	100 %



Mode 854.25 µm **Span** (d90-d10)/d50 51.02 **D[4,3]** 258.7 µm

Q3(x) [%]	x [µm]
5	1.1
10	**2**
25	6.7
50	**18.2**
75	648.1
90	**930.9**
95	1°052.9
99	1°264.3

2 A

1 C

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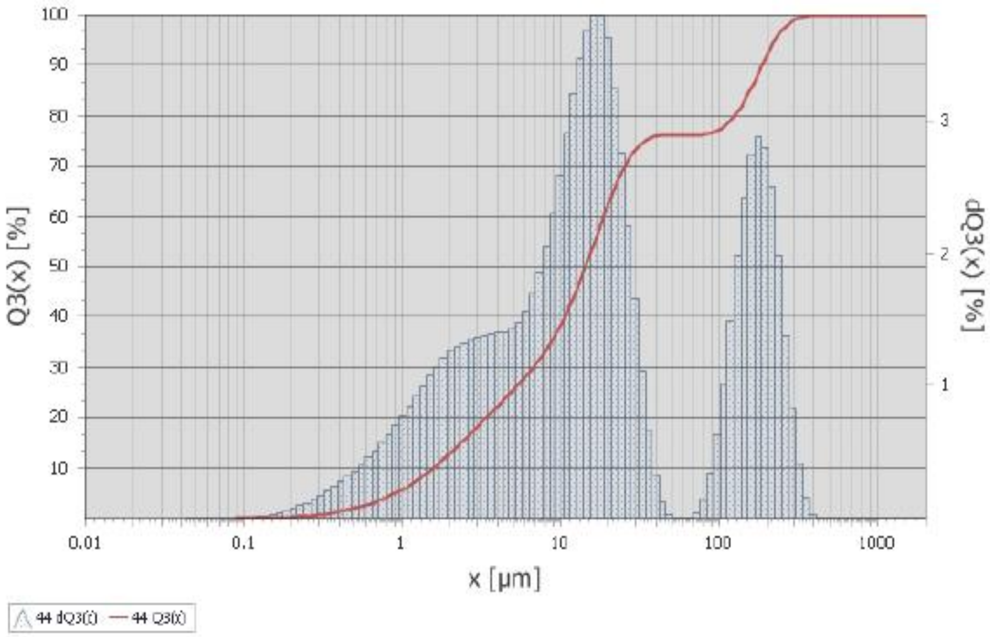
ANALYSETTE 22 **MicroTec plus**

FRITSCH

Meas.No.	44	**Date**	mercoledì, 26 ottobre 2016 03:40
Attr. 1		**Attr. 2**	
Attr. 3		**Attr. 4**	

Calculation	Fraunhofer			**TradeOff**	broad (1000.0)		
Refractive Index	---			**Absorption Index**	---		
Scans Fine	100	**Scans Coarse**	100	**Channels**	102	**Beam Obscuration**	11.0 %
Meas. Range	0.08 [µm]	-	2000.00 [µm]	**Pump**	60 %	**Ultrasonics**	100 %



Mode 17.33 µm **Span** (d90-d10)/d50 12.88 **D[4,3]** 51.9 µm

Q3(x) [%]	x [µm]
5	0.9
10	**1.6**
25	4.9
50	**14.4**
75	34.3
90	**186.9**
95	225.5
99	292.9

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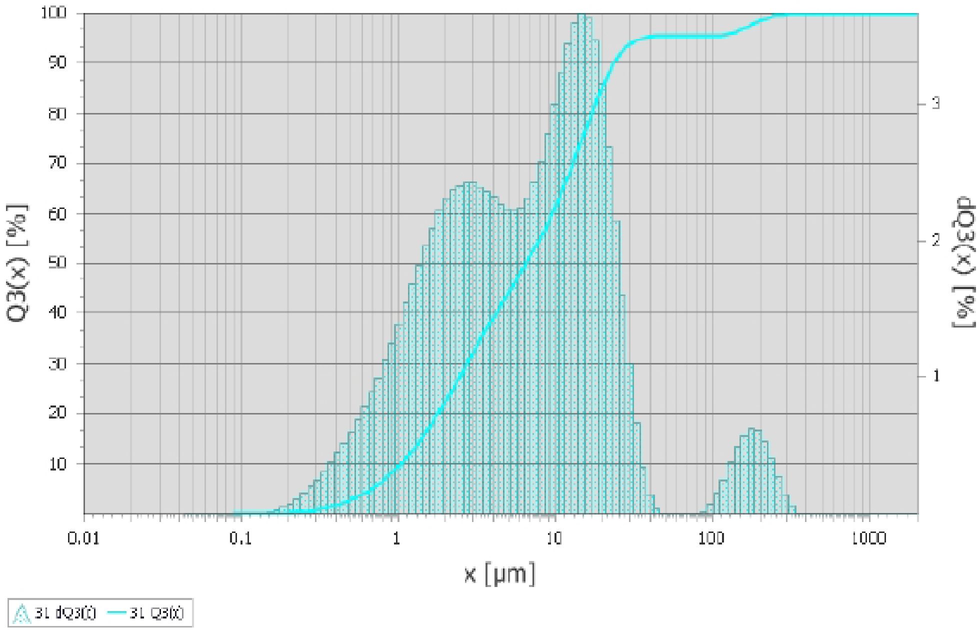
ANALYSETTE 22 MicroTec plus FRITSCH

Meas.No.	31	**Date**	mercoledì, 26 ottobre 2016 01:32
Attr. 1		**Attr. 2**	
Attr. 3		**Attr. 4**	

Calculation	Fraunhofer		**TradeOff**	broad (1000.0)	
Refractive Index	---		**Absorption Index**	---	
Scans Fine	100	**Scans Coarse** 100	**Channels** 102	**Beam Obscuration**	16.0 %
Meas. Range	0.08 [µm] - 2000.00 [µm]		**Pump** 60 %	**Ultrasonics**	100 %



Mode 15.22 µm **Span** (d90-d10)/d50 3.48 **D[4,3]** 16.6 µm

Q3(x) [%]	x [µm]
5	0.7
10	**1**
25	2.2
50	**6.5**
75	14.9
90	**23.7**
95	34.9
99	221.9

2 B

2 C

ANALYSETTE 22 **MicroTec plus**

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Meas.No. 39 **Date** mercoledì, 26 ottobre 2016 03:06

Attr. 1 **Attr. 2**

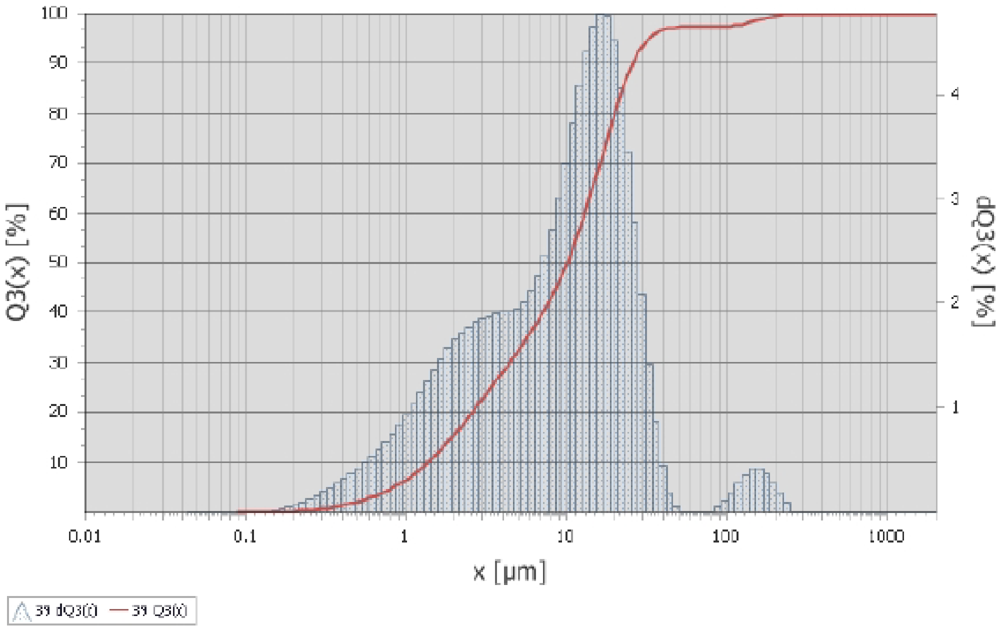
Attr. 3 **Attr. 4**

Calculation Fraunhofer **TradeOff** broad (1000.0)

Refractive Index --- **Absorption Index** ---

Scans Fine 100 **Scans Coarse** 100 **Channels** 102 **Beam Obscuration** 11.0 %

Meas. Range 0.08 [µm] - 2000.00 [µm] **Pump** 60 % **Ultrasonics** 100 %



Mode 16.78 µm **Span** (d90-d10)/d50 2.43 **D[4,3]** 15.2 µm

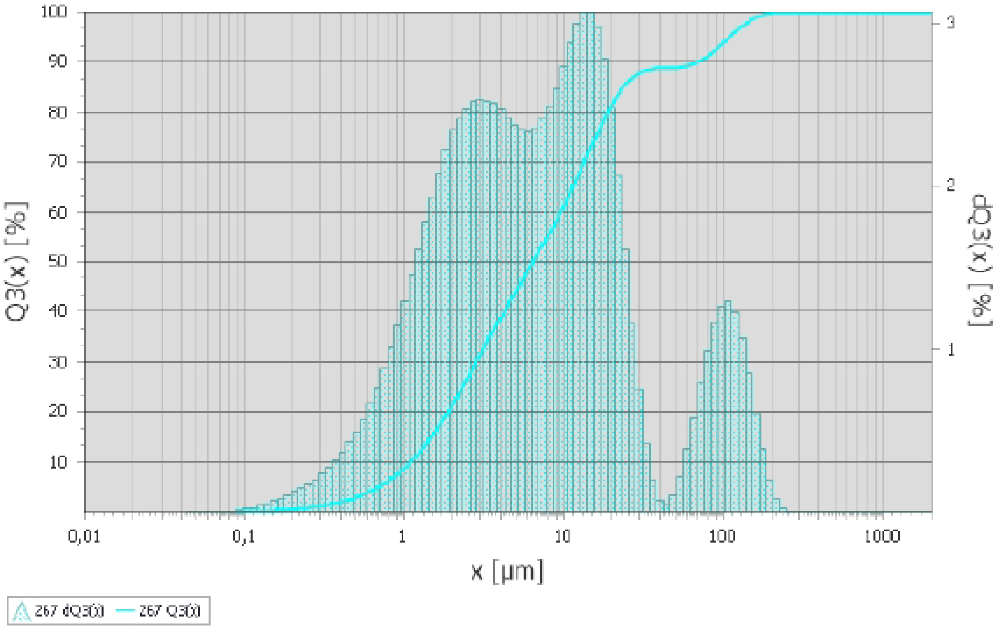
Q3(x) [%]	x [µm]
5	0.8
10	**1.4**
25	3.4
50	**10.1**
75	18
90	**26**
95	32.5
99	164.7

3 A

ANALYSETTE 22 MicroTec plus

Meas.No.	267	**Date**	giovedì, 26 gennaio 2017 10:25
Attr. 1		**Attr. 2**	
Attr. 3		**Attr. 4**	

Calculation	Fraunhofer		**TradeOff**	broad (1000,0)	
Refractive Index	---		**Absorption Index**	---	
Scans Fine	100	**Scans Coarse** 100	**Channels**	102	**Beam Obscuration** 12,0 %
Meas. Range	0,08 [µm] - 2000,00 [µm]		**Pump**	60 %	**Ultrasonics** 100 %



Mode 13,81 µm **Span** (d90-d10)/d50 10,41 **D[4,3]** 19,1 µm

Q3(x) [%]	x [µm]
5	0,7
10	**1,1**
25	2,3
50	**6,4**
75	15,9
90	**67,8**
95	108,4
99	160,1

3 B

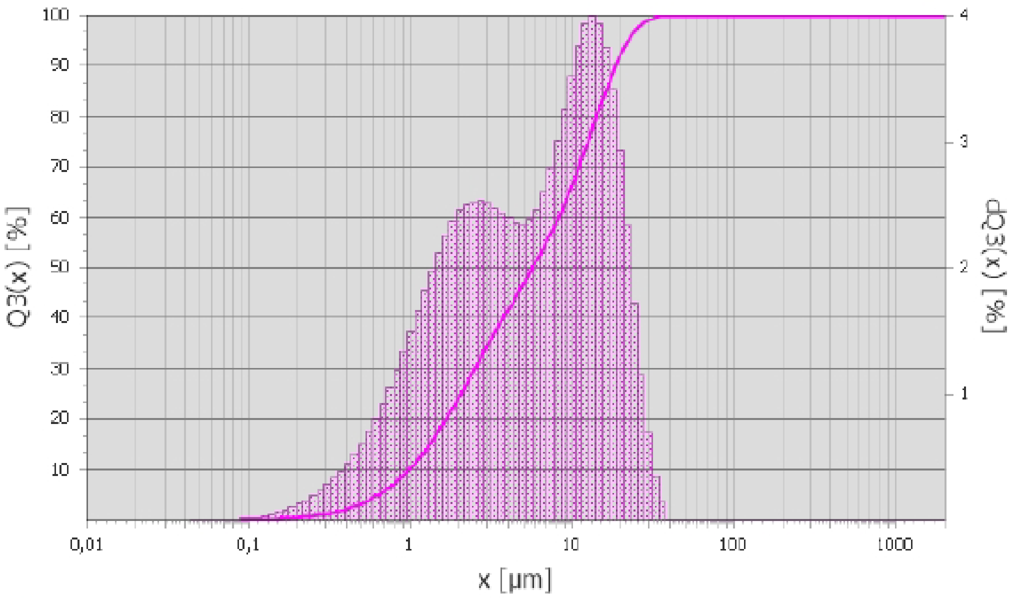
ANALYSETTE 22 **MicroTec plus**

Meas.No.	81	**Date**	martedì, 13 dicembre 2016 05:03
Attr. 1		**Attr. 2**	
Attr. 3		**Attr. 4**	

Calculation	Fraunhofer	**TradeOff**	broad (1000,0)
Refractive Index	---	**Absorption Index**	---

Scans Fine 100 **Scans Coarse** 100 **Channels** 102 **Beam Obscuration** 12,0 %

Meas. Range 0,08 [µm] - 2000,00 [µm] **Pump** 60 % **Ultrasonics** 100 %



Mode 13,36 µm **Span** (d90-d10)/d50 3,08 **D[4,3]** 8,0 µm

Q3(x) [%]	x [µm]
5	0,6
10	**1**
25	2,1
50	**5,7**
75	12,5
90	**18,6**
95	22,2
99	28,9

3 C

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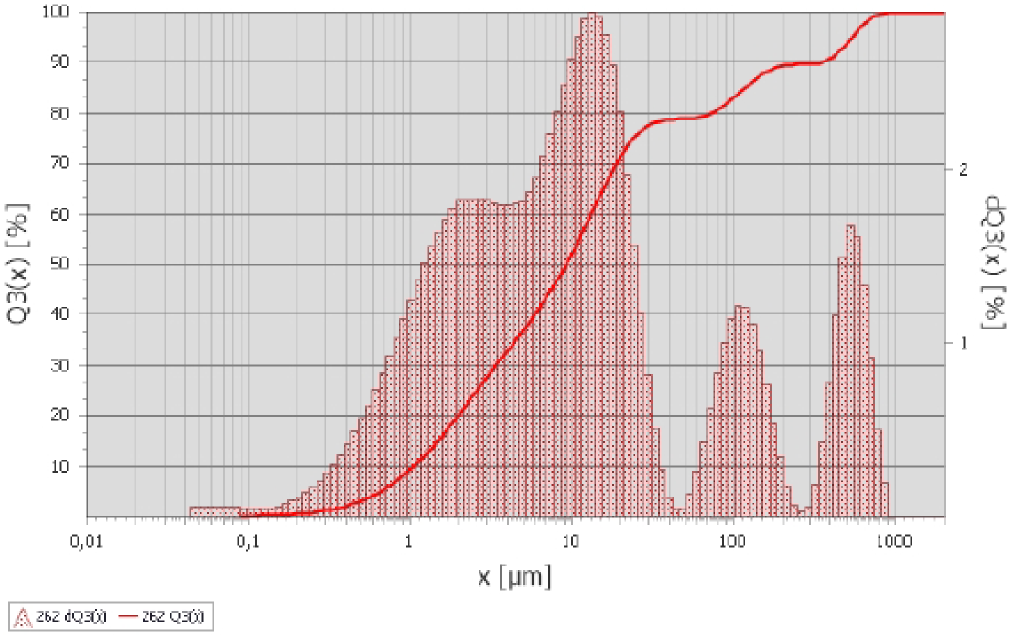
ANALYSETTE 22 MicroTec plus

Meas.No.	262	**Date**	giovedì, 26 gennaio 2017 09:59
Attr. 1		**Attr. 2**	
Attr. 3		**Attr. 4**	

Calculation	Fraunhofer	**TradeOff**	broad (1000,0)
Refractive Index	---	**Absorption Index**	---

Scans Fine	100	**Scans Coarse**	100	**Channels**	102	**Beam Obscuration**	14,0 %
Meas. Range	0,08 [µm]	-	2000,00 [µm]	**Pump**	60 %	**Ultrasonics**	100 %



Mode 13,36 µm **Span** (d90-d10)/d50 37,33 **D[4,3]** 75,5 µm

Q3(x) [%]	x [µm]
5	0,7
10	**1,1**
25	2,6
50	**9,2**
75	24,1
90	**343,5**
95	540,8
99	723,6

4 A

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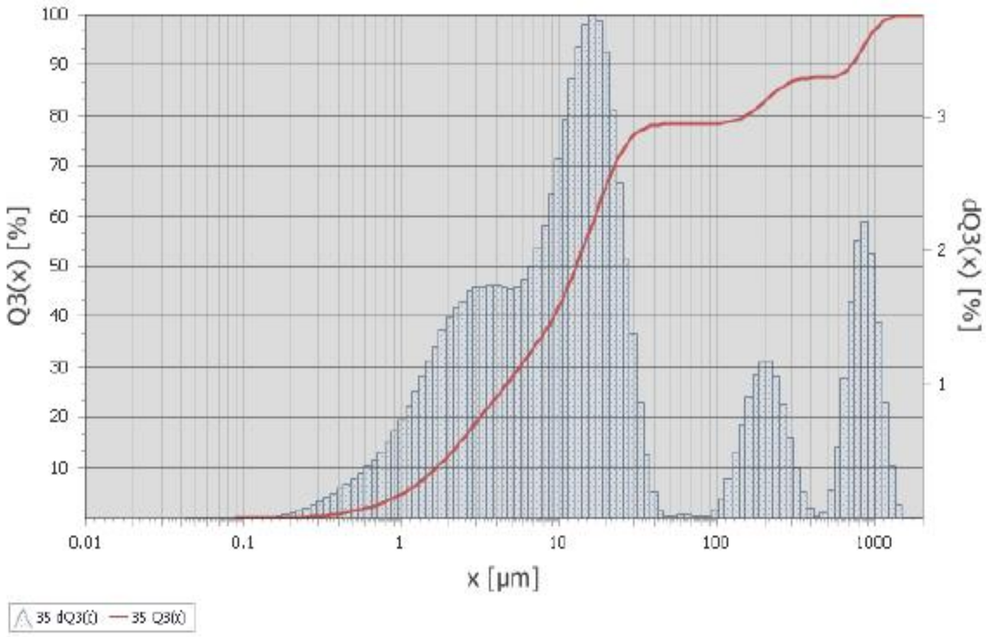
ANALYSETTE 22 MicroTec plus

FRITSCH

Meas.No.	35	**Date**	mercoledì, 26 ottobre 2016 02:46
Attr. 1		**Attr. 2**	
Attr. 3		**Attr. 4**	

Calculation	Fraunhofer			**TradeOff**	broad (1000.0)		
Refractive Index	---			**Absorption Index**	---		
Scans Fine	100	**Scans Coarse**	100	**Channels**	102	**Beam Obscuration**	10.0 %
Meas. Range	0.08 [µm] - 2000.00 [µm]			**Pump**	60 %	**Ultrasonics**	100 %



Mode 16.78 µm **Span** (d90-d10)/d50 55.01 **D[4,3]** 136.7 µm

Q3(x) [%]	x [µm]
5	1
10	**1.7**
25	4.2
50	**12.9**
75	27.8
90	**708.4**
95	904.1
99	1°159.9

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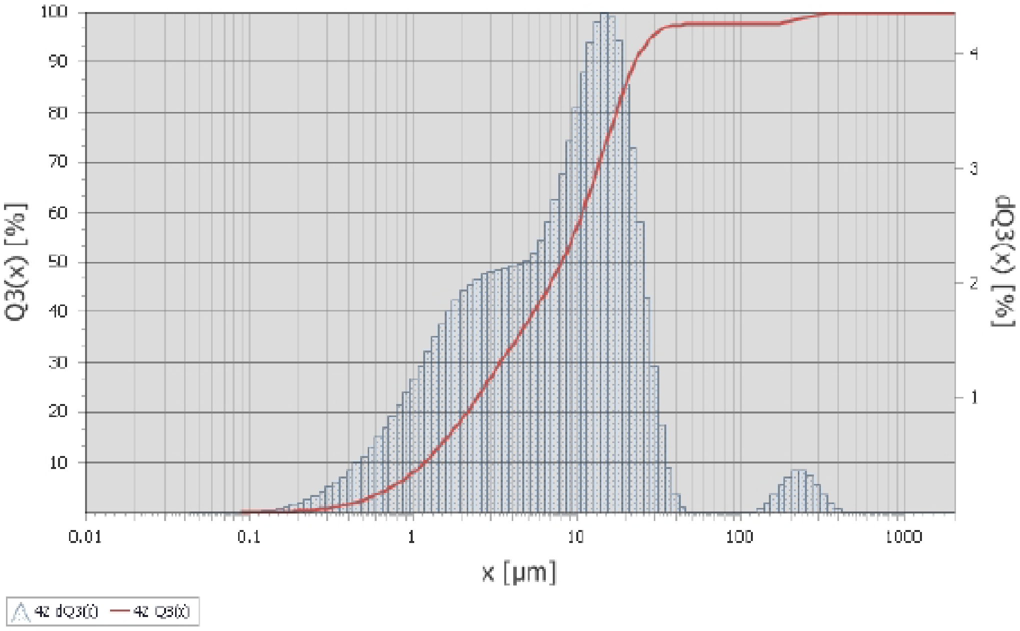
ANALYSETTE 22 MicroTec plus

Meas.No.	42	**Date**	mercoledì, 26 ottobre 2016 03:29
Attr. 1		**Attr. 2**	
Attr. 3		**Attr. 4**	

Calculation	Fraunhofer	**TradeOff**	broad (1000.0)
Refractive Index	---	**Absorption Index**	---

Scans Fine	100	**Scans Coarse**	100	**Channels**	102	**Beam Obscuration**	11.0 %
Meas. Range	0.08 [µm]	-	2000.00 [µm]	**Pump**	60 %	**Ultrasonics**	100 %



Mode 15.22 µm **Span** (d90-d10)/d50 2.68 **D[4,3]** 14.9 µm

Q3(x) [%]	x [µm]
5	0.7
10	**1.2**
25	2.7
50	**8**
75	15.4
90	**22.7**
95	28.1
99	239.9

4 C