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Comparing Long-Term Observations of Sediment Yield with Estimates of Soil Erosion Rate Based on Recent ^{137}Cs measurements. Results From an Experimental Catchment in Southern Italy

Paolo PORTO[1,2*], Giovanni CALLEGARI[2]

[1]Dipartimento di Agraria, University Mediterranea of Reggio Calabria, Feo di Vito, 89122, Reggio Calabria, Italy

[2]Institute for Agricultural and Forest Systems in the Mediterranean (ISAFOM) - National Research Council (CNR), Via Cavour, 4/6, 87036, Rende (CS), Italy

E-mail: paolo.porto@unirc.it

*Corresponding author

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Abstract

Recent increase of soil erosion rates in Southern Italy emphasizes the need to identify areas subjected to higher environmental risks. Over the last few decades, several techniques have been used to this purpose. They include mainly experimental sites, like plots and catchments of different size, and the use of prediction models calibrated using local parameters. More recently, some of the limitations associated with these techniques suggested the use of radiotracers, mainly ^{137}Cs, both as alternative and complementary tools to traditional methods. However, some of the assumptions associated with the application of the ^{137}Cs technique still require testing and validation to provide robust estimates of soil erosion rate. These assumptions relate mainly the ^{137}Cs spatial variability on the reference area and the effects of possible additional fallout due to nuclear accidents (e.g. Chernobyl and/or Fukushima). In this contribution, a small experimental catchment located in Calabria, Southern Italy, was selected as a study area by virtue of its long-term record (29 years) of sediment yield

measurements. This database made it possible a comparison between long-term observations and estimates of soil erosion rate provided by recent measurements of ^{137}Cs. The overall results indicate that if the uncertainty of the reference value is taken into account and the Chernobyl additional flux is incorporated into a physically based conversion model, the latter is able to provide robust estimates of soil erosion rate in the area.

Keywords: Soil erosion, ^{137}Cs, Model validation, Experimental catchments, Italy

1. INTRODUCTION

Soil erosion by water affects large areas of forested and cultivated lands of Southern Italy. Experimental sites, mainly plots and catchments of different size, documented peaks of soil loss higher than 100-150 t ha^{-1} yr^{-1} if associated with exceptional rainfall events (see Porto et al., 2018; 2022). Recent studies carried out in Calabria indicated also a general increase in the number of heavy precipitation events, intensity and frequency since about 1950 (Capra et al. 2017). These effects have important consequences for environmental risks such as floods, landslides, land degradation, and pose serious questions on their economic impact on soil productivity for the coming years. Recent investigations made at larger scale by Panagos et al. (2018) indicated that the annual cost of loss in agricultural productivity is estimated at ca. €1.25 billion in Europe. Italy is one of the countries that suffers the highest economic impact especially in those areas where human activities are based mainly on agriculture. In order to establish effective countermeasures to contain such problem and to set the priorities of intervention, it is necessary to identify the areas at higher risk. These actions require robust techniques able to provide suitable estimates of soil erosion rate. To date, a number of different techniques have been used to measure and predict soil erosion and sedimentation rates in different areas of Italy. Among them, traditional monitoring techniques, such as experimental plots or catchments, are essential to monitor soil erosion over different land-use and topography (see Bagarello et al., 2018) but are expensive and time consuming; the use of empirical (Bagarello et al.,

2011) or theoretically based models (Ferro and Porto, 2000) proved to be very useful, if locally calibrated, but they have certain limitations in terms of validation at larger scales and over long-term periods. Alternative approaches, based on the use of fallout radionuclides (FRNs), including mainly cesium-137 (^{137}Cs), proved to be very effective to assemble information on long-term and spatial patterns of erosion and deposition rates especially if coupled with existing traditional methods (Porto et al. 2014; Porto and Walling, 2015). The fallout radionuclide ^{137}Cs (with a half-life of ca. 30 years) is associated with the testing of nuclear weapons occurring from the late 1950s to the early 1960s (Ritchie and Ritchie 2005; IAEA, 2014; Zapata et al., 2002). The successful application of this technique is based on a comparison between the inventory values obtained in single sampling points established in the area under investigation and a so-called '^{137}Cs reference value' obtained in undisturbed sites at time of sampling. For several decades the technique was applied in many areas of the world and demonstrated many advantages that include in turn:

1) the relative simplicity of the approach (Zapata et al., 2002; IAEA, 2014);
2) the possibility, after a single field visit, to obtain retrospective information of soil redistribution rates (Mabit et al., 2008; Walling and Quine, 1992);
3) the ability to get a spatial distribution of the estimates derived from single sampling points (Mabit et al., 2009; Zhang et al., 2019);
4) the opportunity to calibrate and validate empirical and theoretical soil erosion models based on both lumped and distributed approaches (Walling and He, 1998; Porto and Walling, 2015; Zhang et al., 2017);
5) the potential to avoid additional costs due to the establishment of experimental plots or catchments (Loughran, 1989);
6) the ability to provide sediment budgets for large areas especially if sedimentation sites like floodplains and reservoirs are surveyed and periodically monitored (Campbell et al., 1988; Gellis and Walling, 2011; Belyaev et al., 2013; Minella et al., 2014; Navas et al., 2014).

There are, however, some important limitations associated with this technique. These are as follows:

1) the estimates of soil redistribution rates are related to a reference value that establishes the basic line of ^{137}Cs in the area. In the absence of direct measurements of fallout, this value is deduced from soil sampling on stable sites located in the vicinity of the study area. This choice assumes that no disturbance has occurred in those sites since the commencement of fallout in 1954 (see Zapata et al., 2002), and that the spatial variability of the ^{137}Cs inventory is limited (see Sutherland, 1994; 1996). These assumptions are sometimes overlooked, especially in areas affected by additional ^{137}Cs fallout (like Chernobyl and Fukushima accidents), and this can lead to biased estimates of the ^{137}Cs reference value and, consequently, erroneous estimates of erosion and deposition rates;

2) the choice of a conversion model able to convert loss or gain of ^{137}Cs into estimates of soil erosion or deposition needs attention. To date, a number of conversion models have been used in many areas of the world (see Walling et al., 1999; 2005). These include (a) empirical equations able to establish a relationship between soil erosion rates and ^{137}Cs inventory losses (see among the others Ritchie and McHenry, 1990; Elliott et al., 1990; Walling and Quine, 1990; Loughran and Campbell, 1995), and (b) theoretical approaches (different for cultivated and uncultivated sites) that try to interpret the variation of ^{137}Cs activity along a soil profile (Yang et al., 1998; Walling and He, 1999; Porto et al., 2003). Basically, these models need calibration and validation using local parameters and this limitation is often overlooked or ignored in many studies;

3) most of the estimates of erosion rate provided by the ^{137}Cs technique are essentially unvalidated (Porto et al., 2003). The main reason is related to the lack of long-term measurements of soil loss available for a time window comparable with that covered by the ^{137}Cs measurements. Only a few contributions made in this direction are available in the literature and include (a) the work of Zhang (2018), in which 34 years of soil loss measurements, carried out since 1978 in a cultivated catchment unit (1.6 ha in size) in USA, were compared with equivalent estimates obtained from ^{137}Cs; (b) the works of Porto et al. (2004; 2014; 2018), in which sediment yield

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measurements covering between 17 and 24 years obtained in 3 uncultivated catchments (ca. 1.5 ha in size) in southern Italy were used for the same purpose; (c) a shorter-term, but significant, study made by Kachanoski (1987) in Canada, in which 11-year measurements over ten experimental plots (44-m long) were used to test the ^{137}Cs re-sampling approach. Other attempts made in this direction are available in the literature but, to the authors knowledge, they are limited to short-term observations of soil erosion and this makes such comparison inappropriate (see, among the others, Evans et al., 2017; Boardman and Evans, 2020; Evans and Boardman, 2021).

In the light of the above considerations, it is clear that the basic assumptions required by the ^{137}Cs technique need to be further validated in contexts where long-term measurements of soil erosion are available and where information on ^{137}Cs spatial and temporal variability can be deduced.

In this paper, a small experimental catchment located in Southern Italy, for which a 29-year dataset of sediment yield observations is available, was used for this purpose. Recent soil sampling campaigns undertaken within the catchment area and over a large reference area allowed to test the ability of a theoretical conversion model to derive soil erosion rates in this area. The overall analysis revealed that if the spatial variability of ^{137}Cs inventory in the reference area is taken into account, and a Chernobyl component is incorporated into the model, the ^{137}Cs technique provides a range of soil erosion rates that compares very well with that covered by the long-term record of sediment yield observed at the catchment outlet.

2 MATERIALS AND METHODS

2.1 The experimental catchment W2

The W2 (1.38 ha in size) is a sub-catchment of the larger Crepacuore basin (18.2 km^2) that drains to the Jonian sea ca. 5 km North of Crotone (39°04’50”N, 17°07’38”E) in Calabria, Southern Italy (**Fig. 1**). The W2 is located at Brasimato (39°07’52”N, 17°01’36”E), at a mean elevation of ca. 98 m a.s.l. and with a mean slope of ca. 35%. Geologically, the area is incised into the Upper Pliocene and

Quaternary clays, sandy clays and sands and shows a soil texture made up of a silt-clay content of ca. 86% (Di Stefano et al., 2005).

The area experiences large seasonal fluctuations in climate with most of the rainfall occurring in winter (from October to March) and concentrated in a few large events that account for more than 50% of the annual value (Porto and Callegari, 2019). Based on the data collected at the near station of Crotone (39°04'45"N, 17°07'00"E), the mean annual rainfall related to the period 1916-2018 is ca. 610 mm while the mean annual air temperature is ca. 17.7 °C.

Figure 1 Study area and location of the experimental catchment W2

The catchment W2 has never been under cultivation and originally it supported a rangeland vegetation cover consisting mainly of *Aegylops ovata*, *Atriplex halimus*, *Lepturus cylindricus* and *Lygeum spartum*. In 1968 the catchment was afforested with Eucalyptus trees (*Eucalyptus occidentalis* E.) in order to 1) reduce soil erosion in this region and b) expand the forested area in Calabria to increase wood production. In 1978, a national 'Project on Soil Conservation' (Porto and Callegari, 2021) was launched to monitor the effects of soil erosion in the country and the catchment W2 was included in this framework. For this reason, this experimental unit was equipped to provide specific information on rainfall, runoff and sediment yield in Southern Italy. More specifically, rainfall and runoff are measured respectively with a mechanical raingauge located in the upper part of the catchment and an H-flume weir (Brakensiek et al., 1979) coupled with a mechanical streamgauge located at the catchment outlet (**Fig. 2**). Sediment yield records are provided using a Coshocton wheel placed downstream the H-flume and coupled with two sized tanks that collect the portion of runoff that is intercepted by the sampler. Measures of rainfall, runoff and sediment yield are generally obtained at event scale. However, when short-time interval occurred between two or more consecutive events, only one measurement of sediment yield was possible and the corresponding data of rainfall and runoff responsible for that sediment output were aggregated accordingly. The sediment output for each event (or cumulated events) is calculated by multiplying the mean sediment concentration measured at the tanks times the corresponding runoff volume. The annual sediment yield is then obtained by the sum of the sediment output related to the erosive events which occurred in that year.

Figure 2 The device to monitor runoff and sediment yield at the catchment outlet

In 1978, at the beginning of the experiment, the eucalyptus trees were cut in order to evaluate the effectiveness of the canopy cover on soil loss. A second cut was made in 1990 in order to 1) monitor the long-term effect of afforestation on soil protection and 2) quantify the wood production of the eucalyptus plantation. From 1990 to date no silvicultural treatments were applied and the condition of trees was maintained more or less stable even if recent tree mortality related to an attack of *Phoracantha semipunctata* (a species of beetle originating from Australia) caused a number of single trees to fall down.

2.2 The sediment output dataset

The measurements of sediment output started at the beginning of the experiment in 1978. Unfortunately, occasional malfunctioning of the equipment caused some interruption of the long-term monitoring and for some years, that include 1981, 1988, 1989, and the period from 1995 to 2005, the full records of sediment yield at the catchment outlet are not available. In December 2005 the sampling equipment was refurbished and the monitoring activity recommenced, with no loss of data until recent days. To date, a long-term (30 years) dataset of sediment yield is available for this catchment and the annual values from 1978 to 2020 are illustrated in **Fig. 3**.

Figure 3 The annual values of sediment yield at the catchment outlet

These measurements indicate a large inter-annual variability of sediment production that ranges from a minimum of 1.7 t ha^{-1} in 1987 to a maximum of 95.9 t ha^{-1} in 1992, with a mean value equal to 22.4 t ha^{-1} yr^{-1} and a standard deviation equal to 25.0 t ha^{-1} yr^{-1}. This variability reflects both the seasonal regime of the rainfall, characterized by the occurrence of frequent flash floods, and the silvicultural treatments applied to the trees that caused the exposure of soil to the rainfall impact. For example, the existence of two extreme outliers in 1990 and 1992, associated with a particularly high value of annual rainfall, coincided with a period of reduced vegetation cover as a result of logging operations in 1990. These outliers emphasise the importance of extreme events to erosion (see Porto and Callegari, 2019) and affected the uncertainty of the mean value provided by the long-term record.

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2.3 The soil sampling campaigns for ^{137}Cs measurements

The application of the ^{137}Cs technique in the W2 catchment was based on a comparison between the inventory values obtained in single sampling points established in the catchment area and a ^{137}Cs reference value obtained at time of sampling over a stable, undisturbed location. More specifically, the ^{137}Cs measurements used in this paper relate to a) 55 sampling points established within the catchment, that served to estimate the mean value of soil erosion in the area from the commencement of fallout (in 1954) to the year of sampling (2021), and b) 12 sampling points selected on a new identified reference area located ca. 7 km distant from the study catchment.

2.3.1 The soil sampling campaign within the catchment area

This sampling campaign was undertaken in the catchment W2 in February 2021, in order to check the hydrological response of the catchment following a severe rainfall event occurred in November 2020. The purpose of the sampling was to collect information on ^{137}Cs inventories within the catchment and to compare each value with the reference values obtained in a stable area. In this campaign, 55 replicate bulk cores were collected following the sampling design established in 1998 by Porto et al. (2001). The samples were taken at the intersections of a 20 m x 20 m grid (**Fig. 4**) in which additional sampling points were considered to cover those areas characterized by marked variability of slope or canopy cover. The cores were collected using a steel core tube (11 cm diameter – 40 cm length) driven into the ground by a motorized percussion corer and extracted by hand. Because very low activity (or absence) of ^{137}Cs was expected for higher depths, each single core was divided into two parts (20 cm in length each). The corresponding layers of the two cores were bulked and the two composite layers were analyzed separately. This strategy served to ensure that a) the ^{137}Cs activity was minimal or absent in the deeper layer and b) the ^{137}Cs activity of the upper layer was high enough to reduce time of analysis.

Figure 4 Location of the sampling points in the catchment W2

During this campaign, additional sectioned cores were also collected from several representative sites within the catchment to provide information on the characteristics of the ^{137}Cs depth profiles within the catchment.

2.3.2 The soil sampling campaign to establish the basic line of ^{137}Cs in the study area

Recognized international protocols for the application of the ^{137}Cs technique (see among the others Kachanoski and de Jong, 1984; Zapata et al., 2002; IAEA, 2014; Porto et al., 2018) suggest that the ^{137}Cs reference value can be estimated (a) from direct fallout data (where available), (b) from ^{137}Cs levels at undisturbed areas (noneroded sites) located in the vicinity of the study area, or (c) from previous measurements carried out at the same site (re-sampling strategy). In alternative, if none of the three above criteria can be applied because fallout data, or previous ^{137}Cs measurements, are not available and undisturbed location cannot be easily identified at short distance, a software provided by IAEA can be used for this purpose (see IAEA, 2014). The software was established for applying conversion models to ^{137}Cs inventories and it includes provision for estimating the local bomb fallout reference inventory at a site, based on its global coordinates and its mean annual rainfall.

To date, direct fallout measurements are still not available near the study area but previous investigations made by Porto et al. (2001, 2003, 2016, 2018) provided the basis for deriving a robust ^{137}Cs reference value to be used in this work. A first ^{137}Cs reference value for catchment W2 was obtained during a campaign undertaken in 1999 (Porto et al., 2001). In that case, a small undisturbed area (10 m^2), covered by permanent grass, was individuated in the vicinity of the study catchment and one sectioned core (surface area of 652 cm^2) plus six additional bulk cores (8.6 cm internal diameter) were collected. The radiometric analysis provided, for the sectioned core, a value of the reference inventory equal to 2609 Bq m^{-2}, and for the six cores a mean value of 2637 Bq m^{-2} with a sd = 315 Bq m^{-2} (values decay correct to 1998). A subsequent work made by the same research group (see Porto et al., 2003) provided two additional reference values equal to 2430 Bq m^{-2} (sd = 272 Bq m^{-2}) and 2445 Bq m^{-2} (sd = 246 Bq m^{-2}) (values decay correct to 2001) obtained, respectively, from a small clearing (9 m^2) between the eucalyptus trees located in an adjacent catchment, and from an undisturbed area (12 m^2), located at the same altitude and covered by pasture with some scattered oaks. The same sampling strategy was adopted with a single sectioned core of larger surface area (652 cm^2) and six additional bulk cores (8.6 cm internal diameter) collected around each site. In 2014, a new sampling campaign was carried out, and the same reference site sampled in 1999 was revisited to obtain a new reference value for ^{137}Cs (see Porto et al., 2018). In this case, eight soil samples from a square area (25 m^2 in size) were collected using the same sampling device but a larger core tube (internal diameter = 11 cm). Each sample was sectioned into increments of 1-4 cm and all the corresponding subsamples from the eight cores were combined. A single bulked sample was obtained for each depth and a total reference inventory of 1796 Bq m^{-2} (value decay correct to 2014) was provided by the radiometric analysis. In this case, even if the sampling strategy allowed to account for the micro-scale variability, no calculation of the standard deviation was possible.

Details of the reference values obtained in the past campaigns described above are reported in **Table 1**. The list includes also the estimate, provided by the IAEA software, of the ^{137}Cs bomb fallout inventory at Crotone (1196 Bq m^{-2}), located ca. 10 km distant. All the values have been standardized to a fixed date at the end of 2020 in order to facilitate the comparison.

When looking at **Table 1**, two main issues can be noticed. The first issue relates to the on-site variability of the mean reference inventories. In other words, even if the mean values obtained in different sampling campaigns are comparable, they show large ranges with values of CV between 11% and 13% due to micro-scale variability. These values are in the acceptable range suggested by other authors to make the cesium technique suitable (see Owens et al., 1996; Sutherland 1996). However, some explanation of this spatial variability should be provided and, more important, this inventory variation should be taken into account to establish to which extent the uncertainty related to the reference values can affect the final estimates of soil erosion obtained by the single points within the catchment area. A second, important, issue relates to the estimate of the ^{137}Cs bomb fallout inventory at Crotone (1196 Bq m^{-2}) provided by the IAEA software. This is lower than the measured values provided by the soil sampling campaigns and suggests that the area may have received Chernobyl fallout which has resulted in an increased inventory. In this respect, a recent contribution by Porto et al. (2016) reported finding evidence of a Chernobyl peak in some sediment cores collected from a reservoir close to the W2 catchment. They estimated that the total Chernobyl fallout input was small, but this could provide one explanation of the different inventories and their spatial variability reported in **Table 1**. In order to explore further these two issues, another specific sampling campaign was undertaken in March 2021 over a larger stable area (ca. 1 ha in size), in which a systematic collection of soil samples was possible. Unfortunately, the sites in which the previous campaigns were undertaken are not available anymore due to a recent cultivation of the area, and a new, undisturbed location, ca. 3.7 km distant from the study catchment, was selected for this purpose (see **Fig. 5a**). Collection of 12 soil sectioned cores was made at the intersections of a 40 m x 40 m grid (**Fig. 5b**) using the same sampling device employed by Porto et al. (2018) and consisting of a motorized soil column cylinder auger set in which a core tube with internal diameter of 11 cm is accommodated. Each sample was sectioned into increments of 2 cm and was analyzed separately for ^{137}Cs content.

Figure 5 The reference area a) location and b) sampling points

2.4 The Sample Analysis

After collection, all the samples were transported to the University of Reggio Calabria, where radiometric analyses were carried out. Prior to the analyses of ^{137}Cs activity, all samples were air dried and then oven dried for 48 h at 105°C and weighed to calculate the mass depth. After disaggregating and sieving to <2 mm, each sample was mixed for 2h using a 3D shaker to homogenize the ^{137}Cs content. Then, representative sub-samples were placed into Petri dishes or plastic pots of different sizes depending on the sample mass. The radiometric analyses were made by two Canberra p-type HPGe detectors, model GX4020, coupled with a multichannel Desktop Spectrum Analyzer DSA-1000 Canberra. Both detectors are characterised by a relative efficiency of 45.6% with a resolution of 1.1 keV at 122 keV and 2.0 keV at 1.33 MeV. The Canberra Genie 2000 software package was used to perform the spectral analysis. More specifically, a Monte Carlo procedure, associated with the Canberra's LabSOCS (Laboratory SOurceless Calibration Software) code and able to establish the calibration efficiency for any type of geometry was preliminarly performed for each detector. The energy calibration was obtained using a certified ^{155}Eu and ^{22}Na multigamma source with a wide energy range (42.8–1274.5 keV). Subsequently, a validation phase was performed using the activity concentration for ^{137}Cs of standard materials presented to the detectors in containers of identical geometry to those used for the study (Petri dishes or Marinelli beakers). The standards were produced by adding a measured amount of certified liquid standard to a known amount of <2 mm soil with a ^{137}Cs activity below the level of detection and representative of the samples to be analysed. This operation was repeated for each detector and the two types of equipment did not show

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substantial differences in terms of efficiency. The ^{137}Cs activities in the samples were obtained from the counts at 662 keV. The background noise of the detection system was also evaluated using blank samples of identical geometry in order to improve the accuracy of the measurement. All the samples were placed at a distance d = 0 from the top of the detector and the counting time varied from ca. 80,000 to ca. 240,000 s depending on the activity that was expected for each sample. The above procedure provided final results with an analytical precision of ca.10% at the 95% level of confidence. The lower limit of detection depends on efficiency and counting time, and thus, it differs from sample to sample. The inventory (Bq m^{-2}) of each bulk core was calculated as the product of the measured ^{137}Cs activity (Bq kg^{-1}) and the dry mass of the <2 mm fraction of the bulk core (kg), divided by the surface area of the core (m^2). The total inventory corresponding to the sectioned cores (Bq m^{-2}) was obtained in turn by summing the above product (Bq) of all layers, into which the core was sectioned, divided by the surface area of the core (m^2).

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ha eliminato: The samples were measured by two Canberra p-type high-resolution low energy coaxial HPGe detectors (model GX4020) with a relative efficiency of 45%. The detectors are calibrated using the software LABSOCS (Laboratory Sourceless Calibration Software) and a multigamma source. Counting time varied from ca. 80,000 to ca. 240,000 s depending on the activity that was expected for each sample…

3. RESULTS

3.1 The ^{137}Cs inventories at the reference site

The total ^{137}Cs inventories associated with the 12 sectioned cores collected from the reference area in March 2021 are reported in **Table 2**. It is clear from **Table 2** that, even if the calculated mean value (1623 Bq m^{-2}) is consistent with those obtained from the past campaigns (listed in **Table 1**), it resulted higher than that (1196 Bq m^{-2}) estimated at Crotone for the global fallout and indicates a possible Chernobyl flux. Also, a general spatial variability can be observed. The measurements show a sd of 187.9 Bq m^{-2} and a range between 1363 Bq m^{-2} and 1976 Bq m^{-2}, again, consistent with those reported in **Table 1** obtained from the previous analyses. The 12 corresponding ^{137}Cs profiles are illustrated in **Fig. 6** in which the ^{137}Cs activity (Bq kg^{-1}) is plotted against the corresponding mass depth (kg m^{-2}). All profiles conform with those typical of undisturbed sites with a maximum value at the soil surface (Profiles 2, 5, 8, 9, and 10) or a peak a few cm below (Profiles 1, 3, 4, 6, 7, 11, and 12).

Figure 6 ^{137}Cs depth distribution associated with the 12 profiles collected from the reference area. The dashed line indicates the simulation obtained with the DMM (Eq. 3)

3.2 The ^{137}Cs inventories within the catchment W2

The values of ^{137}Cs inventory obtained for the 55 bulk cores collected in February 2021 ranged from 29 Bq m^{-2} to 1157 Bq m^{-2}, with a mean of 355 Bq m^{-2} and a standard deviation of 306 Bq m^{-2}. All values of ^{137}Cs inventory obtained from these cores are less than the local reference inventories reported in **Table 2**. The general depletion of ^{137}Cs inventory is consistent with that obtained in two previous works by Porto et al. (2001, 2014) in which the same analysis for two different time windows (1954-1998 and 1954-2013) was performed. This result suggests that the catchment has been dominated by net soil loss over the period covered by the ^{137}Cs measurements (i.e. 1954-2020) and that the presence of depositional areas is limited even if local depositional phenomena cannot be excluded at the scale of single events (see Porto and Callegari, 2021).

3.3 Converting the ^{137}Cs inventories into values of soil erosion rate

In order to establish the amount of soil eroded from the catchment area it was necessary to convert ^{137}Cs depletion into rates of erosion. This conversion is generally based on the degree of reduction of the measured inventory, relative to the local reference inventory. Considering the uncertainty associated with the reference inventories shown in **Table 2**, and assuming that all the 12 profiles are equally representative of the reference area, we decided to use all 12 inventories as reference values and to treat the resulting estimates as a range of soil erosion rates instead of a single value.

The conversion model used in this analysis is the Diffusion and Migration Model (DMM), in the refined version proposed by Porto et al. (2003). This version of the DMM is based on a simulation of the ^{137}Cs activity along a soil profile following measured records of the atmospheric fallout and its temporal redistribution. Based on this premise, the DMM attempts to reproduce the activity of ^{137}Cs, for a single value of mass depth *x* and time *t'*, with the following equation:

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

$$\left. \times \frac{1}{\sqrt{4\pi D(t-t')}} - \frac{V}{2D} e^{\frac{Vx}{D}}\, erfc \left[\frac{x-E+y+V(t-t')}{\sqrt{4D(t-t')}} \right] \right\} dy \qquad (1)$$

where:

$C\,(x,t,t')$ (Bq kg^{-1}) represents the ^{137}Cs activity simulated for a single value of mass depth x and time t';

$I\,(t')$ expressed in (Bq m^{-2} yr^{-1}) indicates the ^{137}Cs amount of fallout that has reached the ground at time t' (yr);

H (kg m^{-2}) is a constant that represents the ^{137}Cs initial relaxation mass depth;

D (kg^2 m^{-4} yr^{-1}) is a diffusion coefficient, assumed constant in time and space;

V (kg m^{-2} yr^{-1}) is a migration coefficient representing the constant downward migration rate;

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

x (kg m^{-2}) indicates the cumulative mass depth;

t (yr) is the time elapsed since the commencement of fallout in 1954;

E (kg m^{-2}) indicates a constant rate of lowering of the soil surface by erosion (E = 0 for a reference profile);

erfc(u) is the error-function complement defined as (Crank 1975):

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

At first, Eq. (1) is integrated for each value of x using a fixed value of t' and a soil layer of variable thickness y (here expressed in kg m^{-2} as a mass length) for which a diffusional transport in a water saturated porous medium is assumed (Lindstrom & Boersma, 1971; Pegoyev & Fridman, 1978).

Then, in order to get the total value of ^{137}Cs concentration C (x,t) (Bq kg^{-1}) for that value of x, it is necessary to account for the fallout input occurred from 1954 to the date of sampling. This will be obtained by integrating Eq. (1) over time t', assuming a continuous input $I(t')$ viz:

$$C(x,t) = \int_0^t C\,(x,t,t')dt \tag{3}$$

The attempt of Eq. (3) to simulate the diffusion and migration of ^{137}Cs along the soil column was demonstrated in several works carried out in southern Italy (see Porto et al., 2004; 2016; Altieri et al., 2018) and in UK (see He and Walling, 1997).

However, in the present contribution, it was necessary to account for the additional fallout due to Chernobyl accident and the model required the following refinement in respect to the dataset representing the fallout input $I(t')$. The original version of the DMM assumed that the temporal distribution of annual fallout at a study site follows the same relative annual variation as that of a reference station located in the same hemisphere. This means that any difference in absolute magnitude of the local ^{137}Cs deposition flux $I(t)$, can be accounted using the following equation:

$$I(t) = \alpha\, I_n(t) \tag{4}$$

where:

$I_n(t)$ is the ^{137}Cs fallout for the reference station (Bq m^{-2} yr^{-1});

α is a scaling factor that can be calculated as follows:

$$\alpha = \frac{A_{ref}}{\int_{1954}^{t} I_n(t')e^{-\lambda(t-t')}dt\prime} = \frac{A_{ref}}{A_n} \tag{5}$$

where A_n (Bq m^{-2}) is the present total atmospheric fallout inventory for the ^{137}Cs deposition at the reference station (see Walling et al., 2005).

Figure 7 ^{137}Cs fallout in Italy (mean values for the country) from 1954 to 2020 (from ISIN, 2021 – Replotted)

In the absence of specific fallout measurements in the study area, it was decided to assume as reference fallout inventory the record published by ISIN (2021) that reports the mean ^{137}Cs deposition flux in the country. Based on this record, replotted in **Fig. 7**, a mean value of ca. 6000 Bq m^{-2} can be assumed as Chernobyl-related ^{137}Cs fallout in 1986. This peak is followed by a sharp decrease until 1994 when the radiation levels have rapidly returned to normal. In order to adapt this record to the local study area, a scaling factor α was first calculated from Eq. (5) using as A_{ref} the value (reported in **Table 1**) estimated by the IAEA software (1196 Bq m^{-2}), and as A_n the value estimated at national scale in the absence of Chernobyl contribution (this value, decayed to 2020, resulted ca. 3011 Bq m^{-2}). The calculation of α (= 0.397) served to rescale the bomb fallout and allowed the calculation of the Chernobyl-related ^{137}Cs fallout for each of the 12 profiles obtained in the local reference area. These values, expressed as total Chernobyl fallout in 1986 (Bq m^{-2}) and as a percentage to the total reference inventory in 2020, are reported in **Table 2**.

Based on the above assumptions, the ability of Eq. (3) to fit the measured reference profiles used in this paper is shown in **Fig. 6** where the dashed line representing the model is superimposed on each experimental profile.

In order to use Eq. (3) to get the estimate of soil erosion rates from each sampling point, it is necessary to integrate it over mass depth *x*. This integration will give the total ^{137}Cs inventory A_u (Bq m^{-2}) for an erosion site at time *t*:

$$A_u(t) = \int_0^\infty C\,(x,t)dx \tag{6}$$

Eqs. (1), (3), and (6) can be solved simultaneously for E (kg m^{-2}), by replacing A_u (Bq m^{-2}) with the measured inventory related to an eroding point. Erosion rates R (kg m^{-2} yr^{-1}) may then be obtained by dividing the quantity E by the time t-t_0 (yr) elapsed from the commencement of ^{137}Cs fallout (1954) to the date of sampling. The results of this application exercise are reported in **Table 2** in which the value of erosion rate corresponding to each profile is related to the average of the erosion rate obtained for the 55 sampling points in the catchment.

4. DISCUSSION

The histograms illustrated in **Fig. 7** and the comparison between measured and simulated ^{137}Cs profiles depicted in **Fig. 6** provide a first important issue of discussion about the ability of the DMM to reproduce the ^{137}Cs distribution along a soil profile in areas affected by the Chernobyl fallout. The first, visual, inspection of the twelve graphs in **Fig. 6** indicates a certain flexibility of this model if the additional input related to the Chernobyl accident is incorporated into the calculation routine. As stated above, local ^{137}Cs flux is not available for the study area, but the reconstruction of the annual fallout by combining measurements at national scale and local inventories obtained in stable areas, seems to be convincing. In fact, looking at the values reported in **Table 2**, the estimates of ^{137}Cs fallout in 1986 range between 356 Bq m^{-2} and 1623 Bq m^{-2} with a mean value of ca. 894 Bq m^{-2} and a standard deviation of ca. 8.7 Bq m^{-2}. These values are in line with those provided by the 'Atlas of caesium deposition on Europe after the Chernobyl accident' (see De Cort et al., 1998) in which ^{137}Cs values ranging from 0 to 1 kBq m^{-2} are indicated for Calabria (see Table 13 of that report). A previous report published by Izrael et al. (1996) provided a general map of the country that appears to show the possibility of a Chernobyl fallout of between 400 and 2000 Bq m^{-2} in Calabria and this is, again, consistent with our estimates at the reference area. Based on these assumptions, the ability of the DMM to simulate the profiles illustrated in **Fig. 6** must be recognized and its use as conversion model can be recommended even in the presence of significant Chernobyl input. The second, important, issue of discussion is related to the rates of soil erosion provided by the application of the DMM in the study catchment. Looking again at the values reported in **Table 2**, the estimates of soil erosion rate provided by the twelve profiles in **Fig. 6** and related to the period covered by the ^{137}Cs measurements (1954-2020), range between 15.6 t ha^{-1} yr^{-1} and 23.6 t ha^{-1} yr^{-1} with a mean value of ca. 20.4 t ha^{-1} yr^{-1} and a standard deviation of 2.7 t ha^{-1} yr^{-1}. These values, expressed as a range to take account of the uncertainty associated with the estimate of the Chernobyl component, are depicted in **Fig. 8** together with the measurements of sediment yield obtained at the catchment outlet. The values of measured sediment yield show a very large range, as reported above. For this reason, it was convenient to express these measurements as a range around the central value (given by the average and its uncertainty assumed at the 95% level of confidence, i.e. ± 2SE). Also, in order to quantify the incidence of the Chernobyl input on our experimental site, we reported in **Fig. 8** the corresponding estimates of erosion rate using the DMM without the additional 1986 fallout. This calculation provided a range of soil erosion rates between 14.8 t ha^{-1} yr^{-1} and 21.0 t ha^{-1} yr^{-1} with a mean value of ca. 18.9 t ha^{-1} yr^{-1} and a standard deviation of 2.3 t ha^{-1} yr^{-1}.

Figure 8 Comparison between net erosion rates provided by ^{137}Cs measurements and values of sediment yield for two different periods

It is clear from **Fig. 8** that the estimates of net erosion provided by the ^{137}Cs measurements, with and without the Chernobyl component, fall within the range covered by the values of sediment yield measured at the catchment outlet during the observation period (from 1978 to 2020) if the uncertainty related to the sampling device is taken into account. It is also clear that, if the Chernobyl component is considered, the range of estimates shows a better overlapping with the sediment yield data while the estimates provided by neglecting the 1986 input divert to lower values.

The mean value of net soil loss derived from ^{137}Cs (20.4 t ha^{-1} yr^{-1}) cannot be considered significantly different of that (22.4 t ha^{-1} yr^{-1}) measured at the catchment outlet and related to the monitoring period.

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These findings suggest that the ^{137}Cs technique is a useful means to provide soil erosion estimates if the uncertainty related to the reference value is considered. However, a perfect agreement between these two sets of values cannot be expected because they cover two different time windows: the ^{137}Cs measurements are related to the period comprised from the commencement of fallout (1954) to the date of sampling (2021), while the measurements of sediment yield were obtained for a shorter period covered by the catchment monitoring (1978-2020). A logical comparison would have required the availability of measurements at the catchment outlet for the same period covered by the ^{137}Cs measurements (1954-2020). Unfortunately, such dataset in our study area is not available but detailed information on rainfall erosivity and the use of a calibrated prediction model can help to derive suitable estimates of sediment yield for this longer period. More specifically, available information on the rainfall erosivity factor *R* (Wischemeier and Smith, 1978) for the period 1954-2020 (Porto, 2016) and details on soil, topography and land use, obtained for the study catchment (see Di Stefano et al., 2005), allowed the application of the SEDD model (Ferro and Porto, 2000) to estimate of sediment yield for the missing years of the same period. The SEDD model was already calibrated and validated for the catchment W2 (see Ferro and Porto, 2000; Porto et al., 2018) and it provided reliable estimates of sediment yield based on long-term observations. The results of this simulation indicate a mean value of sediment yield equal to 19.2 t ha^{-1} yr^{-1} with a range (± 2SE) comprised between 14.6 t ha^{-1} yr^{-1} and 23.8 t ha^{-1} yr^{-1}. These estimates, plotted in **Fig. 8** for comparison, are surprisingly close to those provided by the ^{137}Cs measurements and confirm, once again, the ability of the ^{137}Cs technique to provide suitable estimates of erosion rates in similar contexts.

5. CONCLUSIONS

Recent measurements of ^{137}Cs, carried out over an experimental catchment in Southern Italy, provided important information on the ability of the ^{137}Cs technique to estimate soil erosion rates in this area. The work reported here suggests how the Diffusion and Migration Model (DMM), able to convert ^{137}Cs loss into values of soil loss, can be used to get robust estimates of soil erosion rate even in presence of a Chernobyl additional flux. For this purpose, it was necessary to select a large reference area where the spatial variability of ^{137}Cs inventory due to the Chernobyl input was first investigated, and to incorporate this additional input into the calculation routine of the DMM. The estimates of soil erosion rate were treated as a range instead of a single mean value, and were compared with the long-term observations of sediment yield (29 years) obtained at the catchment outlet. This comparison showed a very close agreement between measured and estimated values of soil erosion and encourages the use of the ^{137}Cs technique in different geographic contexts.

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