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ORIGINAL ARTICLE

Investigation of on-site implications of tea plantations on soil erosion in Iran using ^{137}Cs method and RUSLE

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Abstract

Tea plantations cover an area of ca. 31,000 ha in the Central North of Iran. This area, after clearing the original forests more than 50 years ago, became exposed to severe soil erosion. The objective of this study was to assess long-term soil erosion rates in tea farms and to evaluate their soil conservation effect at a whole-of-catchment scale. No previous information on direct measurements of soil erosion in this mountainous tea agro-ecosystem of Northern Iran is available. Point-based estimates of net erosion have been obtained using the ^{137}Cs technique and these results were compared with estimates using the RUSLE model. Calculations of soil erosion rates from ^{137}Cs inventories, based on the Mass Balance Model II, revealed that 1.3 mm year^{-1} and 1.45 mm year^{-1} of soils were lost from the two sub-catchments A1 and A2, into which the catchment can be divided. The annual net erosion rate of the entire catchment was 17.06 t ha^{-1} year^{-1} which is consistent with the rate of 20.4 t ha^{-1} year^{-1} obtained by the RUSLE model of at the same scale. This study suggests adopting soil conservation practices to control soil erosion, especially after deforestation and periodic pruning of tea bushes. Sustainable land-use plans are then necessary for tea farms of Iran to protect soil resources and to reduce the off-site impacts of land degradation (mainly siltation) on the reservoirs and coastal area.

Keywords Deforestation · ^{137}cs technique · Soil erosion · Tea farm catchment (TFC)

Introduction

Land-use changes including deforestation and land clearing play an important role in soil degradation and soil losses worldwide (Gharibreza et al. 2013a, b). One of the most common types of land-use changes in tropical and subtropical areas is the conversion of natural forest to tea farms. In Asia, millions of hectares of original forests were converted to tea farms and they occupy mostly steep slopes where erosion hazard is very high. In Iran, this practice is very common (and is backdated to 110 years), especially along the coast of the Caspian Sea where the subtropical climate creates ideal conditions for tea cultivation. At a national level, the area covered by tea farms increased dramatically during the second half of the past century, from an area of about 11,000 to 31,000 ha (almost triple) between 1952 and 1971 (Yazdani 2009). Since then, the distribution of tea farms has been relatively stable, with an average production of ca.193 tons of green leaves each year. Tea production is dominated by small family farms in Iran. More than 71% of tea farms occupy an area smaller than one hectare and the remaining 29% of farms range from 1 to 5 ha (TOI 2016). Similar land owner structures of tea plantations exist in Sri Lanka where all tea farms are smaller than 4 ha (Herath 2001). In Iran, ca 70% of tea farms (22,030 ha), most of them more than 50 years old, were established on steep slopes (10 to 30%)

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and this exposes them to high soil erosion hazards. Renovation is an essential and periodic procedure on tea farms, in which the stems of old shrubs will be cut for rejuvenation of plants. This operation increases erosion risk by exposing bare ground to water-induced erosion. Sometimes, the renovation process takes longer because of the low income, and therefore, bare lands remain exposed to erosive agents for long periods. Previous studies (Othieno 1975; Gholiev 1995; Ananda 1998; Hartemink 2006; Emanuelsson and Rasmusson 2012) have demonstrated that soil erosion is an important natural process in these areas, mainly induced by the cultivation methods of tea plants, and the periods of renovation and harvesting. Land degradation involving severe erosion, loss of nutrients, and mass movements of soil and rock debris is also well documented in tea plantations around the world (Ananthacumaraswamy et al. 2003; Yan et al. 2003; Hewawasam 2010; Mupenzi et al. 2011; Zheng et al. 2012; Iori et al. 2014; Chit et al. 2017). In Iran, few attempts have been made to estimate the magnitude of soil erosion in tea plantations, and a reliable technique to cover this issue is still under evaluation. Similarly, specific studies based on special crops such as vineyards, citrus, apricots, and persimmons have been conducted to introduce issues raised by land-use changes, soil erosion, and relevant soil conservation techniques (Rodrigo-Comino et al. 2018; Keesstra et al. 2016). Such studies actually point to the importance of investigating soil erosion based on specific crops.

During the past decades, on-site and off-site impacts of land degradation encouraged researchers to study soil redistribution through erosion plots and empirical models (Sivapalan 1983; Krishnarajah 1985). More recently, the use of nuclear techniques offers great potential in this field (Zhang et al. 2003; Du and Walling 2011; Hamzah et al. 2014; Sahoo et al. 2016; Porto et al. 2016). The application of Cesium-137 (half-life of ca. 30 years) proved to be very useful as a tracer to estimate long-term soil redistribution rates. Its application to tea farms in Iran is reported here for the first time.

Cesium-137 (^{137}Cs) is mainly nuclear weapon test-derived radionuclide (bomb-derived ^{137}Cs) and to a smaller extent nuclear power plant accident-derived radionuclide (Chernobyl ^{137}Cs). It emits gamma rays with an energy of 661.6 keV (Poreba 2006). This radionuclide was first applied by Yamagata et al. (1963) and Rogowski and Tamura (1970) to estimate rates of soil erosion in small areas. Analytical methods and models for estimating soil erosion using ^{137}Cs have improved remarkably over the past four decades (Robbins 1978; IAEA 1995, 1998; Walling and Quine 1990; Walling and He 1999; Zapata and Agudo 2000; Porto et al. 2003; Poreba 2006). Recently, comprehensive handbooks providing detailed guidance on this method are available (Fulajtar et al. 2017; Mabit et al. 2014; Zapata 2002). For conversion of ^{137}Cs inventories to soil erosion rates, a set of conversion models was developed and a PC-compatible Excel- Add developed by Walling et al. (2014, 2007) is available and allows the application of improved models to estimate erosion and deposition rates in both cultivated and non-cultivated areas. Mabit et al. (2008) have evaluated the performance of different conversion models and pointed out the advantages and limitations of using ^{137}Cs for assessing soil erosion. The use of this technique requires an appropriate radiometry laboratory for the measurements and a particular skill to be applied in different situations.

Traditionally, the application of empirical models like the Universal Soil Loss Equation (USLE) (Wischmeier and Smith 1978) and its revised versions including RUSLE (McCool et al. 1994) and USLE-M (Bagarello et al. 2018) are the most widely applied empirical models for estimating soil loss because the lack of input data hinders broader use of more sophisticated process-based models. USLE-based models are applicable for cropland, disturbed forestland, rangeland, and other land uses where rainfall and its associated overland flow cause soil erosion (Cohen et al. 2005; Chen et al. 2012; Bera 2017; Le Roux et al. 2005; Prasannakumar et al. 2011; Zisheng and Luohui 2004). Several studies have compared empirical models like RUSLE and nuclear techniques in estimations of soil erosion in afforested and deforested lands (Ritchie and McHenry 1990; Jimena et al. 2012; Busacca et al. 1993; Di Stefano et al. 2005; Porto and Walling 2015).

However, a literature review proves that the ^{137}Cs method has not been applied in Iran to estimate soil erosion on tea farms, and such results and their results have never been compared with RUSLE. The use of the ^{137}Cs method for validation of USLE-based erosion models is more promising than the use of small erosion plots because the ^{137}Cs method provides more representative long-term erosion rates and a better understanding of the spatial distribution of erosion. The objective of this research was, therefore, to determine the soil redistribution pattern on tea farms within a small catchment typical of the tea cultivation area of Iran using the ^{137}Cs method and RUSLE model and to compare the data obtained by both methods.

Materials and methods

The study area

According to the overall situation of the distribution of tea farms in the North of Iran and the research assumptions, the study area with a catchment scale approach was selected. Prerequisite data have proved deforestation and land-use change to tea farm plantations in the study area. The tea farm catchment (TFC) is located in the Alborz Mountain Range, Northern Iran (coordinate limits are 37°

10′ 52″–37° 11′ 07″ N and 50° 02′ 36″–50° 02′ 52″ E) (Fig. 1). The TFC (14.75 ha in size) was originally covered by the Hirkani forest (deciduous trees), which has been entirely converted to tea plantations under gradual land-use changes since 1963. The catchment is located at the east of Lahijan in the Guilan Province with the highest point located at an elevation of 190 m above sea level. The research divided the TFC into two sub-catchments A1 and A2 to differentiate separately the rate of erosion westward and eastward slopes and to estimate the effects of the direction of precipitation on the study area. A digital elevation model (DEM), developed to prepare a slope map of the catchment, shows that westward slopes range between 42 and 31%, and eastward slopes range between 22 and 28%.

The study area has a semi-humid climate with maximum rainfall occurring in October and a minimum in July. The mean annual precipitation and temperature are 1209 mm and 15.8 °C, respectively (IRIMO 1986). Field observations and laboratory analyses showed that the soil textures of the study area are mainly loam, silty loam, and silty clay loam. These soils developed on undifferentiated metamorphic rocks (phyllites, schists, and quartzites) of late Cretaceous age which have contacted Neogene intrusive rocks. Magmatism of unknown age is a common geological phenomenon of the study area in which granite and granodiorite cause contact metamorphism. The study area is confined between two thrust faults where Cretaceous rock units overlie on Neogene intrusive rocks.

Sampling campaigns for ^{137}Cs measurements

Estimates of soil redistribution rates using the ^{137}Cs technique can be derived by comparing the ^{137}Cs inventory at individual sampling points with a reference inventory obtained from a site representing the local fallout input in a stable area, which has not been affected by erosion or deposition.

The choice of a reliable reference site to establish the ^{137}Cs base level is a crucial point in the application of the technique. In Iran, previous studies carried out in different geographic areas (Gorji et al. 2004; Khodadadi et al. 2018) documented ^{137}Cs reference values ranging from 2190 to 2380 Bq m^{-2}. These values are in line with what would be expected for bomb fallout and no contribution from the Chernobyl nuclear accident was detected in those areas. In Guilan Province, where our catchment is located, previous studies documented additional Chernobyl-derived fallout of the order of magnitude of ca. 50–60% of the total fallout (Vahabi-Moghaddam and Khoshbinfar 2012). In order to account for this effect, two different sites located in Guilan Province were selected. Both sites consist of open areas with a minimum slope (0°–2°) that do not show any evidence of erosion or deposition. The closest reference site to the study area was located at AlisRud mountain, 500 m far from the TCF. Incremental depth sampling was carried out in both locations. A scraper plate with a cutting edge and having a rectangular metal frame (surface area = 1292.71 cm^2) served to collect 2 sectioned core samples (one for each site) at

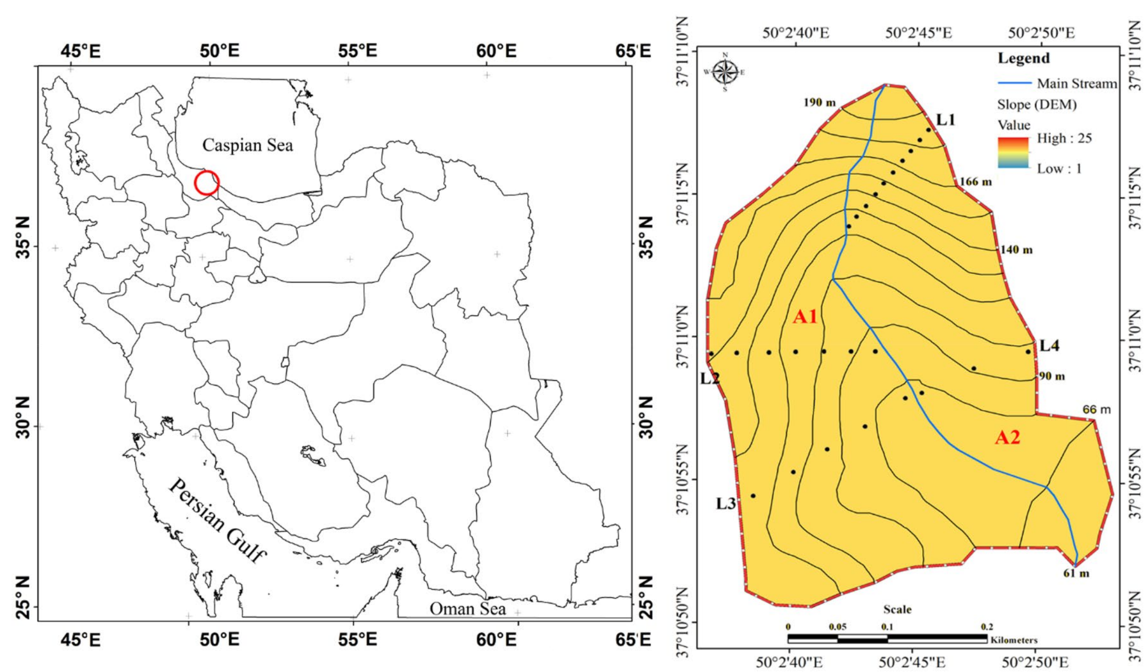

Fig. 1 The geographic location of TFC and sampling pattern (transects)

2-cm intervals to a total depth of 45 cm (to ensure that all the radionuclide was included in the soil profile). No ^{137}Cs activity was detected below 28 cm.

Twelve additional 11 cm-diameter soil cores were also collected from these reference sites, to account for micro-scale variability problems (Owens and Walling 1996). Sampling in the study area consisted of 25 bulk samples collected from four transects following different directions (Fig. 1). In this case, a core sampler of 11 cm-diameter was employed to a depth of 35 cm. Also, one sample from mobilized sediment at the catchment outlet (where deposition occurred) was collected to obtain information on the particle size correction factor (p).

Sample preparation and analyses

The samples were dried at 105 °C, homogenized, weighed, and ground and sieved (2 mm) before analysis. Sub-samples were packed in special containers of 250 g capacity and analyzed for ^{137}Cs content. The ^{137}Cs activities were measured using a well-calibrated gamma-spectrometer based on high-purity germanium (HPGe) detectors in the laboratories of the Nuclear Science and Technology Research Institute (NSTRI), Iran. The gamma-spectrometer model EGPC 80-200-R, (EURISYS MESURES) detector at NSTRI had a relative efficiency of 80% and full width at half maximum (FWHM) of 2.5 keV for ^{60}Co gamma-energy line at 1332 keV. The gamma-spectrometer was calibrated using multi-nuclides standard (POLATOM MIX SOURCE) solutions dispersed in soil homogeneously in the same sample-detector geometry (250 g Marinelli beaker). The lower limit of detection depends on efficiency, FWHM, counting time and so it differs from sample to sample. IAEA reference materials were used for calibration and quality control of the gamma-detector.

A total of 64 samples included sub-samples of sectioned core and single bulk samples were analyzed for finding out grain size distribution and dry bulk density values. The distribution of coarse and fine particle size portions was estimated using the ASTM 422 standard and the ASTM D7928 methods, respectively. Classification of soil texture was carried out using the USDA standard (Roe and Bennett 1927). The dry bulk density of samples was calculated by dividing the dry weight of samples on the volume of bulk corer and reported in kg m^{-3}. Besides, to estimate the land degradation induced by deforestation, the total soil organic carbon (SOC) was measured using the Walkley and Black (1934) method.

The conversion model to estimate soil redistribution rates from ^{137}Cs measurements

Several conversion models able to convert ^{137}Cs loss (or gain) into values of soil erosion (or deposition) are available in the literature (Walling et al. 1999). The study area is affected by the Chernobyl fallout. Therefore, the Mass Balance Model 2 (MBM2) was used (Walling et al. 2014). This model takes into account both the temporal variation in ^{137}Cs fallout input and the initial distribution of fresh fallout on the surface soil. It also distinguishes between bomb-derived and Chernobyl ^{137}Cs fallout (Chernobyl contribution from 5 to 80% of the total amount).

The equation for the MBM2 can be written following the form proposed by Walling and He (1999) in Eqs. 1 and 2:

$$A(t) = A(t_0)\, e^{-\int_{t_0}^{t} (PR/D+\lambda)\,\mathrm{d}t'} + \int_{t_0}^{t} (1-\Gamma) I(t')\, e^{-(PR/D+\lambda)(t-t')}\mathrm{d}t', \tag{1}$$

where: R = erosion rate (kg m^{-2} year^{-1}); D = cumulative mass depth representing the average plough depth (kg m^{-2}); λ = decay constant for ^{137}Cs (year^{-1}); $I(t')$ = annual ^{137}Cs or $^{210}Pb_{ex}$ deposition flux (Bq m^{-2} year^{-1}); Γ = percentage of the freshly deposited ^{137}Cs fallout removed by erosion before being mixed into the plow layer; P = particle size correction factor; t_0 (year) = year when cultivation started; $A(t_0)$ (Bq m^{-2}) = ^{137}Cs inventory at t_0.

If $A(t)$ is greater than the local reference inventory A_{ref} at a sampling point, deposition may be assumed. In this case, the mean soil deposition rate R' can be calculated using the following equation:

$$R' = \frac{\int_{t_0}^{t} R' C_d(t') e^{-\lambda(t-t')}\mathrm{d}t' \int_S R\mathrm{d}S}{PP' \int_{t_0}^{t} \mathrm{d}t' \int_S \left(I(t')\gamma(1-e^{-R/H})/R + A(t')/D \right) \mathrm{d}S}, \tag{2}$$

where: H is the relaxation mass depth of the initial fallout input; $C_d(t')$ reflects the radionuclide content of sediment mobilized from all the eroding areas that converge on the aggrading point. Generally, $C_d(t')$ can be assumed to be represented by the weighted mean ^{137}Cs activity of the sediment mobilized from the upslope contributing area S (m^2); P' is a further particle size correction factor reflecting differences in grain size composition between mobilized and deposited sediment; γ is the proportion of the annual fallout susceptible to removal by erosion before incorporation into the soil profile by tillage.

According to field observations, a tillage depth D = 0.25 m has been assumed. The following values were used for the other input parameters: H = 4, γ = 1 (based on the relationship between the timing of cultivation and the rainfall regime), calculated P factor. A proportion of 60% due to the additional input from Chernobyl was also considered in the overall calculations.

The annual rate of soil redistribution based on percentage reduction in total ^{137}Cs inventory

According to the radionuclide techniques assumptions, ^{137}Cs fallout inputs have been completely mixed within the plough or cultivation layer and the soil loss is directly proportional to the reduction in the ^{137}Cs inventory since the beginning of ^{137}Cs accumulation or the onset of cultivation (Eq. 3).

$$\text{Soil loss\%} = \left(\frac{A_{\text{ref}} - A}{A_{\text{ref}}}\right) \times 100, \tag{3}$$

where A_{ref} = local ^{137}Cs reference inventory (Bq m^{-2}); A = measured total ^{137}Cs inventory at the sampling point (Bq m^{-2}).

The next step is the conversion of percentage reduction in total ^{137}Cs inventory in single bulk samples or an average of reduction along the transect should be converted to the thickness of soil loss. For instance, a 50% reduction in total ^{137}Cs inventory of a sample with the tillage depth of 250 kg m^{-2} and bulk density 1000 kg m^{-3} can be calculated to be 125 kg m^{-2}. Accordingly, the annual rate of soil redistribution can be obtained by dividing the thickness of soil loss on the elapsed time between the beginning of ^{137}Cs accumulation or the onset of cultivation.

The revised universal soil loss equation (RUSLE)

Soil loss for each transect was estimated using the following variant of the USLE equation (Wischmeier and Smith 1978):

$$\mathrm{A} = R \times K \times LS \times C \times P, \tag{4}$$

where A (t ha^{-1} year^{-1}) is the soil loss; R (MJ mm ha^{-1} h^{-1}) is the rainfall erosivity factor; K (t ha h ha^{-1} MJ^{-1} mm^{-1}) is the soil-erodibility factor; L (dimensionless) is the slope-length factor; S (dimensionless) is the slope-gradient factor; C (dimensionless) is the cropping-management factor, and P (dimensionless) is the conservation practice factor.

The R-factor of the study area was assumed to be 1300 (Mj·mm ha^{-1} h^{-1} year^{-1}) based on the rainfall erosivity map of Iran (Nikkami and Mahdian 2014). The K-factor was calculated from clay, silt, very fine sand, sand, and organic matter measured at each sampling point. Permeability and soil structure were estimated based on field measurements and soil texture as well. The obtained K-values range between 0.016 and 0.045 (t ha h ha^{-1} MJ^{-1} mm^{-1}). The LS-factor was obtained using the following relations (McCool et al. 1987; Renard et al. 1994):

$$LS_i = \left(\frac{\lambda_i}{22.13}\right)^{m_i} \left(16.8 \sin \alpha_i - 0.5\right) \quad (\text{valid for } \tan \alpha_i \geq 0.09), \tag{5}$$

where α_i and λ_i are, respectively, the slope angle and the slope length of the unit, and the slope length exponent m_i is given by the following equation (McCool et al. 1989):

$$m_i = \frac{f_i}{1 + f_i}, \tag{6}$$

where f_i represents the ratio of rill to inter rill erosion and can be expressed as follows:

$$f_i = \frac{\sin \alpha_i}{0.0896 \left(3 \sin^{0.8} \alpha_i + 0.56\right)}. \tag{7}$$

The C factor was estimated to be 0.09 based on measurements of canopy and ground cover by tea leaves in the field. Considering that tea plantations have been implemented in a contour line pattern, the conservation factor P was assumed to be equal to 0.45. The model was applied using the conceptual framework reported in Fig. 2 (Bera 2017).

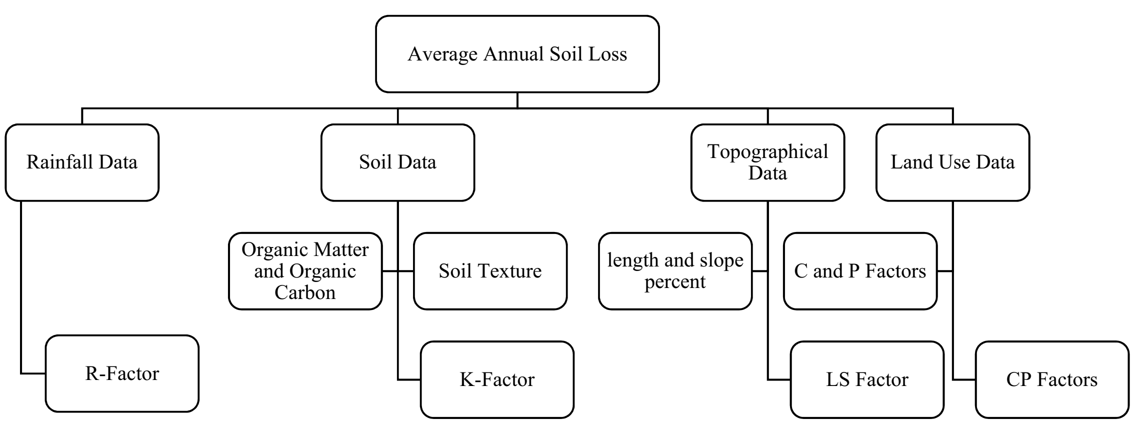

Fig. 2 The conceptual framework for the RUSLE application (After Bera 2017)

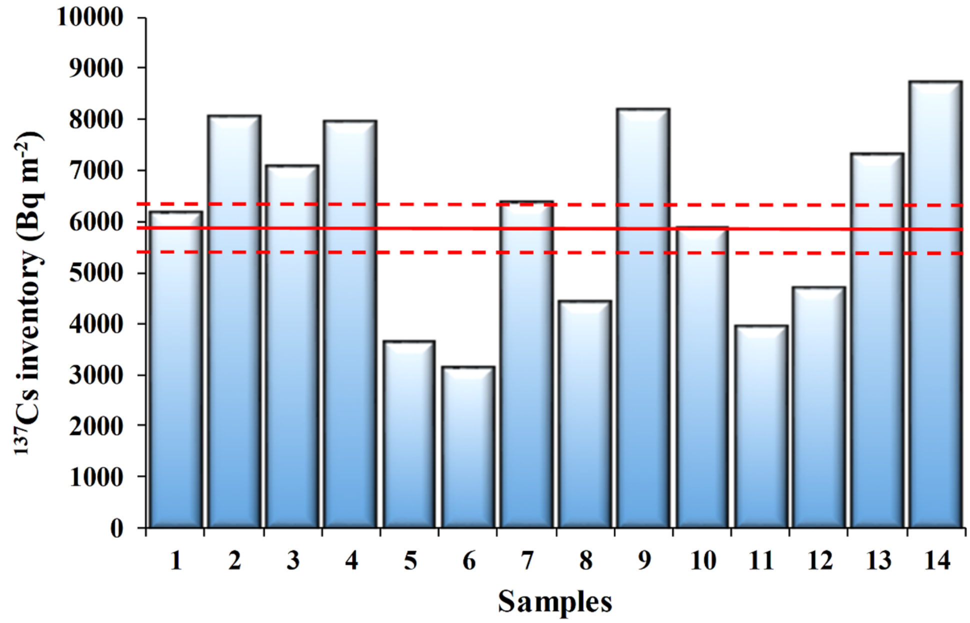

Fig. 3 ^{137}Cs inventory values for 14 samples collected in the reference areas. The straight lines indicate the bootstrap mean and the 95% confidence limits

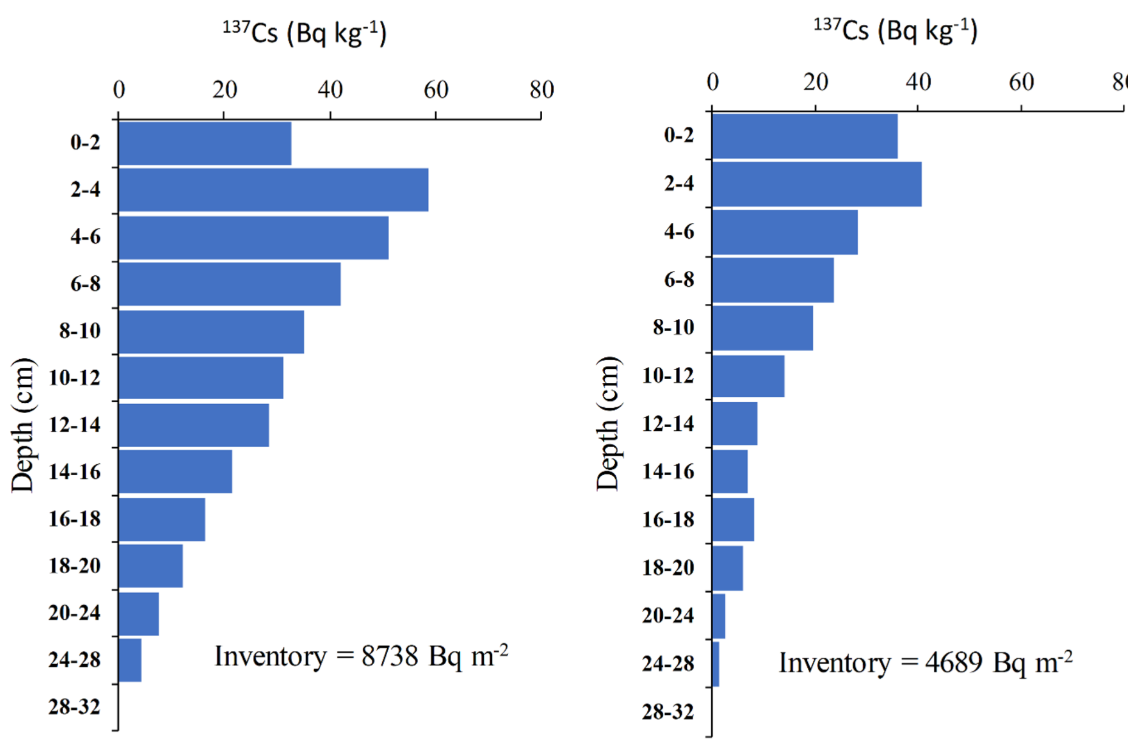

Fig. 4 ^{137}Cs depth distribution from the sectioned cores collected at two reference sites

Results and discussion

^{137}Cs inventories in the reference sites and within the study catchment

The 14 samples collected in the two reference sites located in the Guilan Province provided inventory values ranging from 3134 Bq m^{-2} to 8738 Bq m^{-2} (Fig. 3).

The ^{137}Cs depth distribution of the two sectioned cores (sample no. 12 and no. 14 in Fig. 3) collected from the two reference sites (one for each location) is depicted in Fig. 4.

For each profile, the ^{137}Cs activity shows an exponential decline with depth with a maximum below the soil surface. The shape of these profiles conforms to what can be expected for an undisturbed location, showing ca 80–90% of the ^{137}Cs inventory present in the top 14–16 cm. However, their difference in terms of total inventory is relevant. Overall, this reflects the high variability (CV = ca. 31%) associated with the 14 values collected in both reference sites (Fig. 3). Most importantly, if we compare the mean value obtained from the 14 values (ca. 6113 Bq m^{-2}) with other reference values documented in Iran (2190–2380 Bq m^{-2}. Gorji et al. 2004; Khodadadi et al. 2018), the former is much higher. This difference can be partially attributed to the much higher annual rainfall amount (ca. 1209 mm) in the study area than in the other regions (ca. 300 mm) for which the reference inventories are available. However, this difference and the associated spatial variability suggest the presence of a Chernobyl component in the global fallout. This assumption is supported by previous studies carried out in the South Caspian region that showed evidence of this additional input. Vahabi-Moghaddam and Khoshbinfar (2012), for example, documented ca. 50–60% of the total fallout to be attributed to the Chernobyl accident. Their measurements in different areas of the Guilan region provided a mean inventory value of ca. 6200 Bq m^{-2} that is well in line with what was obtained in our measurements. Other works (Brandt et al. 2000; Quelo et al. 2007) documented the deposition of a weak front of the Chernobyl plume in this area around 5 May 1986 and local meteorological stations recorded scattered precipitation in the same period (IRIMO 1986).

Based on this evidence and the spatial variability of the inventory values, the following approach was adopted to choose the final reference value and to consider the influence of Chernobyl in the estimate of erosion rates.

The problem of spatial variability was addressed using a non-parametric approach, based on the resampling (bootstrap) technique (Davison and Hinkley 1997). Previous studies on the application of the bootstrap technique on ^{137}Cs measurements showed that even for high values of CV this approach leads to robust estimates of the mean value and of the 95% confidence limits around the mean (Di Stefano et al. 2000). Following this assumption, a Monte Carlo technique, with replacement, was used to generate a 10,000 empirical series (with $N = 14$), that could be used to establish the sampling distribution of the associated estimates of a mean reference value. The results provided a mean value of 5905 Bq m^{-2} and the 95% bootstrap confidence limits around this mean were estimated to be 5398 Bq m^{-2} and 6371 Bq m^{-2}. The mean value of 5905 Bq m^{-2} was then assumed as the reference value for the area and a 60% proportion of Chernobyl was imposed to run the MBM2.

^{137}Cs inventories within the study area

The ^{137}Cs measurements obtained in the four transects provided a mean ^{137}Cs inventory of ca. 3493.3 Bq m^{-2} with single values ranging from 831.5 to 7055.5 Bq m^{-2} and a CV equal to ca. 50% (Table 1). These results confirm the relatively high ^{137}Cs inventory in the North-West of Iran, even in degraded forest areas, and suggest once again the presence of additional inputs (Chernobyl component). The single values plotted in Fig. 5 suggests a significant trend ($r = 0.5–0.63$; $p < 0.05$) with topography in which ^{137}Cs inventory seems to increase from the ridge crest to the footslope of each transect.

Possible effects of soil particle size distribution and total organic carbon (SOC) were also considered to explain this trend. The analyses showed a moderate negative correlation ($0.1 \leq r \leq 0.48$; $p < 0.05$) between the ^{137}Cs inventory and SOC content from top to toe in the westward transects, while a moderate positive correlation ($r = 0.55$; $p < 0.05$) from top to toe was obtained in the eastward transects.

When the ^{137}Cs inventory in all samples was compared with the soil particle sizes, a weak positive correlation was obtained with the clay-sized particles ($0.1 \leq r \leq 0.35$; $p < 0.05$), and a significant negative correlation ($0.63 \leq r \leq 0.79$; $p < 0.05$) was found with the silt-sized particles. A moderate positive correlation ($0.435 \leq r \leq 0.62$; $p < 0.05$) resulted also between the ^{137}Cs inventory and the coarse component (sand fraction) of soil samples. Overall, a moderate to strong correlation between silt-sized particles and ^{137}Cs inventory was obtained for all transects with different directions.

Combining these results with weather information obtained in the study area, an attempt can be made to explain the role of slope direction in terms of ^{137}Cs adsorption in these soils. Weather information showed that dominant rains (occurring with a ca. 75% frequency) are induced by western humid wind fronts, and this caused higher precipitation on the crest of the western transects than the eastern ones. However, the erosivity of the higher amount of rainfall could be reduced by the higher SOC content in the corresponding

Table 1 ^{137}Cs inventory, soil grain size distribution, SOC content, and soil texture of the selected Tea Farm

Sample ID	Bulk density (kg m^{-3})	Cs-137 (Bq m^{-2})	SOC %	Clay %	Silt %	Sand %	Texture class
L1-1	1583.77	831.48	3.89	16	48	36	Loam
L1-2	1457.30	1639.47	2.08	23	43	34	Loam
L1-3	1343.40	2283.78	3.03	21	50	29	Loam
L1-4	1387.79	2498.02	1.86	23	39	38	Loam
L1-5	1132.34	5944.80	2.63	22	38	40	Loam
L1-6	1362.66	5586.92	2.88	21	34	45	Loam
L1-7	1329.16	3821.34	2.97	20	34	46	Loam
L1-8	1338.38	1606.05	2.95	19	36	45	Loam
L1-9	1531.85	1991.40	1.15	23	46	31	Loam
L1-10	1324.97	7055.49	2.14	19	33	48	Loam
L2-1	1685.95	842.98	1.06	23	63	14	Silt loam
L2-2	1287.29	2091.84	2.38	21	49	30	Loam
L2-3	1217.77	4109.98	1.45	22	42	36	Loam
L2-4	1345.08	3497.20	3.11	28	45	27	Loam
L2-5	1402.03	2804.06	2.11	34	48	18	Silty clay loam
L2-6	1304.87	5023.76	2.74	26	47	27	Loam
L2-7	1113.92	3063.27	3.89	16	46	38	Loam
L3-1	1223.63	2722.58	2.60	27	56	17	Silty loam
L3-2	1070.37	4281.46	3.57	26	47	27	Loam
L3-3	1243.73	4104.32	3.57	21	40	39	Loam
L3-4	1134.86	6724.02	3.77	27	41	32	Loam
L3-5	1442.23	4362.74	2.22	30	55	15	Silty clay loam
L4-1	1268.02	2979.85	2.69	28	53	19	Silty loam
L4-2	1075.39	2554.05	3.56	34	53	13	Silty clay loam
L4-3	1327.49	4911.70	2.87	30	53	17	Silty clay loam

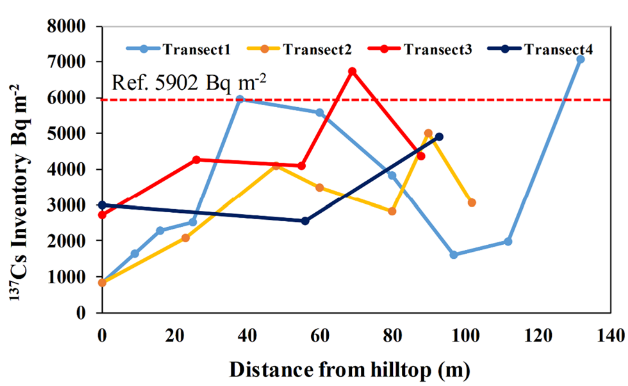

Fig. 5 Distance-dependent from the hilltop of ^{137}Cs inventory and soil erosion in the study area

soils and, as a result, the westward transects are less eroded than the eastern ones. Regarding the enrichment of ^{137}Cs inventory from top to toe of each transect, it is difficult to find a simple explanation. The role of topography is limited because the slope steepness is uniform within the transects and one can expect an increase in soil erosion as the slope length increases. However, the Chernobyl effect may have complicated the ^{137}Cs spatial distribution from top to toe of each transect. As documented elsewhere (Brandt et al. 2000; Quelo et al. 2007), a weak front of the Chernobyl plume arrived in this area from North and North-West directions and it may have caused an enrichment of ^{137}Cs inventories in the areas at the foot-slope of each transect.

Estimating erosion and deposition rates in the sampling points

A PC-compatible Excel-Add in was used to convert the percentage loss or gain in the ^{137}Cs inventory, relative to the local reference value, to a rate of soil loss or deposition associated with the individual sampling points. As explained above, the high variability of the reference inventories (CV = ca. 31%), due to the Chernobyl input, suggests that caution should be exercised in the choice of the reference value. In these cases, as suggested by other authors, the use of a single value can be inappropriate or misleading when it is necessary to estimate rates of soil erosion in large areas (Owens and Walling 1996). As a result, we decided to use as reference inventory a range of values that incorporates the spatial variability measured in the reference areas.

Table 2 Estimates of erosion rate (t ha^{-1} year^{-1}) along the transects using MBM2 (positive values indicate deposition) and RUSLE

Sample ID	Mass balance II	RUSLE
L1-1	− 56.00	− 26.80
L1-2	− 36.00	− 32.10
L1-3	− 26.50	− 33.10
L1-4	− 24.00	− 27.90
L1-5	Stable	− 27.50
L1-6	Stable	− 26.80
L1-7	− 12.10	− 24.20
L1-8	− 36.60	− 24.40
L1-9	− 30.40	− 45.40
L1-10	9.40	− 28.20
L1-MEAN	− 21.20	− 29.70
L2-1	− 55.60	− 42.20
L2-2	− 29.00	− 30.10
L2-3	− 10.10	− 28.20
L2-4	− 14.60	− 19.40
L2-5	− 20.70	− 23.90
L2-6	− 4.60	− 24.20
L2-7	− 18.20	− 31.80
L2-MEAN	− 21.80	− 28.50
L3-1	− 21.50	− 28.70
L3-2	− 9.00	− 30.60
L3-3	− 10.10	− 21.50
L3-4	6.30	− 18.60
L3-5	− 8.50	− 39.20
L3-MEAN	− 8.60	− 27.50
L4-1	− 19.00	− 14.70
L4-2	− 23.30	− 15.60
L4-3	− 5.20	− 13.00
L4-MEAN	− 15.80	− 15.20

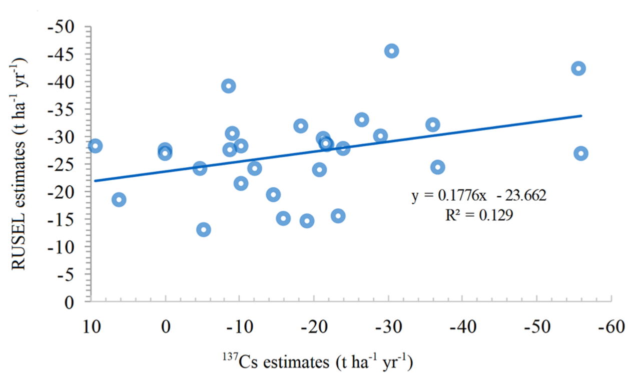

Fig. 6 Comparison between soil erosion estimates using RUSLE and ^{137}Cs measurements

Considering the 95% confidence limits derived from the bootstrap results, the reference inventory was set in the range 5398–6371 Bq m^{-2}. Consequently, the inventory measured in each sampling point is assumed to be greater or less than the reference inventory only when it falls either above or below this range.

The overall results indicated that only two inventory values, obtained from transect 1, were not significantly different from the reference range, indicating that these sampling points were essentially stable. Of the remaining 23 inventory values, 21 were considered to be significantly lower than the reference range, and only 2 were significantly greater. As a final result, 84% of the sampling points had lower values than the reference inventory, indicating that soil erosion (Table 2) was the dominant process since the commencement of ^{137}Cs fallout in the mid-1950s. In Fig. 6, the erosion rates estimated from the ^{137}Cs measurements are compared with the corresponding estimates obtained with the RUSLE model. A weak correlation ($r = 0.36$) between soil erosion estimation of both models was obtained for single bulk samples.

A visual inspection of the graph in Fig. 6 suggests that the RUSLE model provides a general overestimation of erosion rates concerning the corresponding estimates obtained from the ^{137}Cs measurements. The mean values reported in Table 2 for each transect indicate that this overestimation is related to transects 1, 2, and 3 while the mean values for transect 4 are comparable. The perfect agreement would not be expected between the two sets of estimates since the RUSLE model provides only estimates of soil loss and does not account for deposition. However, a possible explanation for this apparent contrast can be related to the different assumptions made by the two models. For example, RUSLE represents erosion induced only by water while the ^{137}Cs method shows an estimation which includes water and wind erosion and tillage effects. More particularly, the ^{137}Cs measurements provide a time-integrated estimate of soil redistribution rates for the period comprised between the mid-1950s to the time of sampling and do not make any assumption on possible changes of canopy and ground cover. On the other hand, RUSLE considers the effect of vegetation but the use of a single value of C (0.09) can be a limiting factor because it does not consider the change in land use which occurred during the 1960s when the natural forest was converted into tea plantations. It should be recognized that tea plantations were absent from the study site for the first 10 years of the period covered by the ^{137}Cs measurements and during that time the area was characterized by a natural forest and by a lower erosion rate.

However, the mean values related to the 25 sampling points taken together, 18.2 t ha^{-1} year^{-1} from ^{137}Cs and 27.2 t ha^{-1} year^{-1} from RUSLE, suggest that soil erosion is a crucial problem in these areas. Based on our measurements, if we assume a mean bulk density value of about ca. 1300 kg m^{-3}, the mean erosion rate during the time between deforestation and sampling is calculated to be ca. 1.4 mm $year^{-1}$ from ^{137}Cs and ca 2.1 mm $year^{-1}$ from RUSLE. These values are not unusual for tea plantations in other geographical areas. Zhang et al. (2003), for example, made a similar experiment in tea plantations of the Southern Jiangsu Province using ^{137}Cs measurements. They found a mean value of soil loss of ca. − 26 t ha^{-1} $year^{-1}$along six transects and this value suggests an erosion rate of ca. 0.22 cm $year^{-1}$. Other contributions that explored the application of the RUSLE model around the world documented estimates of similar magnitude. Chinnamani (1977), in a work conducted in tea plantations established on sloping lands in South India, reported a range of soil erosion estimates as high as − 40 to − 50 t ha^{-1} $year^{-1}$over the years in the absence of any vegetative canopy and soil conservation measure. Equivalent estimates based on RUSLE in tea plantations of the Mahaweli and Nuwara areas in Sri Lanka documented values between − 43 t ha^{-1} $year^{-1}$and − 52 t ha^{-1} $year^{-1}$, respectively (Hewawasam 2010; Abeygunawardena 1993). Other contributions reported in Table 3 suggest the importance of erosion rates in tea plantations. However, it is worth noticing that the authors recognized that the erosion rates provided by RUSLE in the above contributions tend to overestimate water erosion. A similar statement is also reported by Busacca et al. (1993) in Northern Idaho (USA) where RUSLE estimates were three times higher than those provided by the ^{137}Cs technique.

Up-scaling the transect results to the catchment scale

The estimates of soil loss obtained from the four transects provide evidence of soil erosion in tea plantations but they represent on-site values of soil redistribution in the study area and they need to be extrapolated to obtain information at a larger scale. This information represents a key requirement for developing effective sediment management strategies. In order to address this need, the TFC was divided into two different sub-catchments (A1 and A2) (Fig. 1) having an area of 8.84 ha and 5.91 ha, respectively. The extrapolation of the on-site estimates was conducted separately for ^{137}Cs measurements and RUSLE predictions.

The up-scaling of ^{137}Cs estimates was carried out according to the IAEA guideline (IAEA 2014), based on the use of the following equation:

$$E_w = \sum_{i=1}^{n} S_i \times E_i / S_{tot}, \tag{8}$$

where: E_w = Net erosion for the entire catchment (t ha^{-1} $year^{-1}$); n = sub-catchment number; S_{tot} = Surface of the entire watershed or area (ha); S_i = Surface area of the

Table 3 Soil erosion estimations in tea plantations around the world

Location	Slope %	Rainfall (mm)	Erosion (t ha^{-1} $year^{-1}$)	Method of estimation	References
Iran, Guilan, Transect 1	42	1209	− 34.16	^{137}Cs technique, Mass Balance II Model	This study
Iran, Guilan, Transect 2	31		− 33.58		
Iran, Guilan, Transect 3	22		− 18.06		
Iran, Guilan, Transect 4	28		− 23.78		
Iran, Guilan, Transect 1	42	1209	− 34.51	RUSLE	
Iran, Guilan, Transect 2	31		− 34.50		
Iran, Guilan, Transect 3	22		− 28.20		
Iran, Guilan, Transect 4	28		− 15.10		
South India, Nilgiris	10–50	1200	− 40 to − 50	RUSLE	(Chinnamani 1977)
China, Jiangsu Province, B0	0	1385	− 19.46	^{137}Cs technique Mass	Zhang et al. (2003)
China, Jiangsu Province, B1	3		− 27.51		
China, Jiangsu Province, B2	3		− 34.92		
China, Jiangsu Province, B3	3		− 35.41		
China, Jiangsu Province, B4	6		− 15.24		
China, Jiangsu Province, B5	6		− 39.13		
India, South West Ghats	5	1533	− 0.44	RUSLE	Jobin et al. (2017)
Sri Lanka, Mahaweli area	5–30	1500	− 52	RUSLE	Hewawasam (2010)
Sri Lanka, Nuwara Eliya	NA	1500	− 43	RUSLE	Abeygunawardena (1993)
Sri Lanka, Passara	30	2030	− 25.52	Plots	Dharmasena (2011)
Kenya, Kericho, 1-year tea	NA	1060	− 168	Plots	Othieno (1975)
Kenya, Kericho, 2 years' tea			− 81		
Kenya, Kericho, 3 years tea			− 7		
Azerbaijan Republic	15	1800	− 80	Plots	Gholiev (1995)

sub-catchment; E_i = Average net erosion of the representative field(s) of the sub-catchment (t ha^{-1} y^{-1}).

According to Eq. (7), the E_w of the TFC was calculated to be 17 t ha^{-1} $year^{-1}$. It means that total soil loss from TFC during the elapsed time between deforestation and sampling (54 years) has been ca. 922 tons. Annual soil erosion from tea farms of Iran was compared to other crops such as a vineyard, persimmon, and apricot plantations in similar climate conditions in Europe (Rodrigo-Comino et al. 2018; Keesstra et al. 2016; Bayat et al. 2019). Although methods of soil erosion estimation in the reviewed literature were different from the present research, the decreasing order of soil erosion was found to be persimmon > tea > vineyard > apricot plantations, respectively (50 > 17 > 4.1 > 0.91 t ha^{-1} $year^{-1}$).

The application of RUSLE to provide an estimate of net erosion at the catchment scale is more complicated because it requires additional information on sediment delivery. Some authors have suggested (Renfro 1975; Kirkby and Morgan 1980; Walling 1983; Ferro and Porto 2000) that predictions of net erosion at catchment scale can be carried out by coupling a soil erosion model with a mathematical operator that synthesizes the sediment transport efficiency of the hillslopes and the channel network. The latter can be represented by the spatially lumped concept of sediment delivery ratio SDR (Walling 1983) and can be determined only if measurements of sediment yield are available. Unfortunately, direct measurements of sediment yield are unavailable in this catchment and, consequently, SDR cannot be determined using a rigorous procedure. However, some authors provided empirical equations to derive SDR using independent variables (catchment area, slope, particle size), and an attempt to evaluate its magnitude in our study area can be made. Looking at the relationship between SDR and ^{137}Cs measurements in four experimental catchments, ranging in size from 1.5 ha to 32 km^2, Porto and Walling (2015) found a clear positive relationship between SDR and catchment slope. Assuming a mean value of the slope of ca. 30% (derived from the DEM) and using the graph provided by Porto and Walling (2015), a value of SDR of ca. 0.75 can be obtained. If, as an attempt, we multiply this value by the estimate of soil loss provided by the RUSLE at a catchment scale (27.2 t ha^{-1} $year^{-1}$), a final value of 20.4 t ha^{-1} $year^{-1}$ is obtained which is surprisingly close to the estimate of net erosion provided by ^{137}Cs measurements at the same scale. Former guidelines (Schiechti 1985) of FAO and practical methods (Keesstra et al. 2018) have recommended nature-based solutions in agricultural lands and sustainable land management (Visser et al. 2019). The most compatible measures to the tea farms to reduce soil erosion and for enhancing ecosystem services are landscape solutions. Such solutions aim at dis-connecting the water and sediment fluxes when in transport. Keesstra et al. 2018 have recommended some effective management practices such as grassed waterways, vegetation strips, contour planting, and even the use of soil and stone bunds to decrease the sediment delivery of the catchment. These soil conservation practices are vital during periodic rejuvenate tea farms and abandoned farms.

Conclusions

This study represents a unique attempt to obtain estimates of soil erosion in tea plantations using the ^{137}Cs technique at a catchment scale. These estimates are based on a mean reference value of 5905 Bq m^{-2} in which a 60% proportion of Chernobyl-derived radionuclides was assumed. The ^{137}Cs reference inventory of the study area showed that fallout in the North-West forestry areas of Iran (with ca. 1200 mm rainfall) was higher than that measured in other areas of the country characterized by lower values of annual precipitation (ca. 250 mm rainfall).

The results derived from the Mass Balance II conversion model provided similar estimates to those obtained with the RUSLE model and emphasize that MBM2 can be used as a suitable method for evaluating soil loss in areas covered by tea plantations.

However, a high dependency of soil erosion on the gradient and direction of slopes was documented in this study. Overall, the results showed that 105.8 kg m^{-2} (or ca. 7.9 cm, assuming a bulk density of 1343 kg m^{-3}) and 91.1 kg m^{-2} (or ca. 7.1 cm, with a bulk density equal to 1289 kg m^{-3}) of soil has been lost from westward and eastward slopes of the sub-catchments A2 and A1, respectively. The equivalent results obtained for the two sub-catchments A2 and A1 were 1.45 mm $year^{-1}$ and 1.3 mm $year^{-1}$, respectively. These findings showed that erosion is a dominant process in the study area and this is in line with similar results obtained in tea plantations around the world.

The results obtained in this research suggest that the Mass Balance II Model can be used to estimate soil redistribution rates in tea plantations of Iran and, more generally, in other tea farms around the world where deforestation has occurred. The use of alternative models like RUSLE to obtain erosion estimates at a catchment scale should be supported by the ^{137}Cs technique to provide information on sediment delivery. Agronomical and mechanical soil conservation practices to mitigate land degradation should be carried out immediately after deforestation and periodic pruning of tea bushes to minimize soil erosion losses.

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