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Adapting the Caesium-137 technique to document soil redistribution rates associated with traditional cultivation practices in Haiti

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ABSTRACT

Large-scale deforestation, intensive land use and unfavourable rainfall conditions are responsible for significant continuous degradation of the Haitian uplands. To develop soil conservation strategies, simple and cost-effective methods are needed to assess rates of soil loss from farmland in Haiti. The fallout radionuclide caesium-137 (^{137}Cs) provides one such means of documenting medium-term soil redistribution rates. In this contribution, the authors report the first use in Haiti of ^{137}Cs measurements to document soil redistribution rates and the associated pattern of erosion/sedimentation rates along typical hillslopes within a traditional upland Haitian farming area. The local ^{137}Cs reference inventory, measured at an adjacent undisturbed flat area, was 670 Bq m^{-2} (SD = 100 Bq m^{-2}, CV = 15%, n = 7). Within the study area, where cultivation commenced in 1992 after deforestation, three representative downslope transects were sampled. These were characterized by ^{137}Cs inventories ranging from 190 to 2200 Bq m^{-2}. Although, the study area was cultivated by the local farmers, the ^{137}Cs depth distributions obtained from the area differed markedly from those expected from a cultivated area. They showed little evidence of tillage mixing within the upper part of the soil or, more particularly, of the near-uniform activities normally associated with the plough layer or cultivation horizon. They were very similar to that found at the reference site and were characterized by high ^{137}Cs activities at the surface and much lower activities at greater depths. This situation is thought to reflect the traditional manual tillage practices which cause limited disturbance and mixing of the upper part of the soil. It precluded the use of the conversion models normally used to estimate soil redistribution rates from ^{137}Cs measurements on cultivated soils and the Diffusion and Migration conversion model frequently used for uncultivated soils was modified for application to the cultivated soils of the study area, in order to take account of the unusual local conditions. The model was also modified to take account of the fact that cultivation in the study area commenced in 1992, rather than predating the period of weapons test fallout which extended from the mid 1950s to the 1970s. Erosion rates on the upper parts of the hillside involved in the study were found to be relatively high and ca. -23 t ha^{-1} y^{-1} with low spatial variability. In the lower, flatter areas at the bottom of the slope, deposition occurred. Deposition rates were characterized by high spatial variability, ranging from 6.0 to 71 t ha^{-1} y^{-1}. Soil redistribution rates of this magnitude are a cause for concern and there is an urgent need to implement soil conservation measures to ensure the longer-term sustainability of the local agricultural practices.

1. Introduction

The Republic of Haiti occupies the western part of the Caribbean island of Hispaniola (Fig. 1). Agriculture constitutes the main source of livelihood in Haiti. Around 70% of the population depend on this sector, consisting mostly of small-scale subsistence farming (Dolisca et al., 2008). With a fast growing population, expanding from 670 inhabitants per km^2 in 1987 to over 961 in 2013, the demand for arable land has been rapidly increasing at the expense of forests and other areas of native vegetation. Forest cover has reduced from 60% to less

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Fig. 1. Location of the study area, the protected area of the "Forêts des Pins"in Haiti. Photographs of the study site and a typical farmer with his manual tillage tools

than 1% over the last decades (UNDP, 2014; Williams, 2011). Only 20% of the Haitian territory has slopes under 10%, while 63% of the country has slopes above 20% (Dolisca et al., 2007; Ehrlich et al., 1987). As a result of the accelerated deforestation and associated intensive land use on steep slopes, Haiti is now identified as one of most devastated regions in the world in terms of soil erosion (UNEP, 2010). It is currently estimated that only 30% of the Haitian territory is suitable for cultivation and most of this land is exposed to severe soil erosion, resulting in loss of productivity and decreasing crop yields. In addition, crop production is characterized by poor agricultural management practices and a lack of soil conservation measures (UNEP, 2010, 2013).

To address the on-site and off-site consequence of soil erosion in Haiti, there is an urgent need for reliable assessments of the environmental impact of different land uses and associated agricultural practices to inform land managers and decision makers as well as the rural population and to support the design of appropriate mitigation measures. The use of fallout radionuclides (FRNs) has proved to be an effective approach for documenting erosion and sedimentation within various landscapes (Zapata, 2002; Mabit et al., 2013). On reaching the land surface, fallout FRNs are rapidly and strongly adsorbed by the soil and their subsequent redistribution reflects the transport and deposition of soil and sediment particles (IAEA, 2014).

Caesium-137 (^{137}Cs) is a man-made FRN, with a half-life of 30.2 years. It can be found in the soils of the region as consequence of the worldwide atmospheric thermonuclear weapon tests that were undertaken during the period extending from the mid-1950s to the mid-1960s (Walling and Quine, 1995). ^{137}Cs can provide retrospective information on medium-term (50–60 years) soil erosion and sedimentation rates (Zapata, 2002). ^{137}Cs fallout levels were variable globally, reflecting both annual precipitation amount and location relative to the main weapon tests (Walling and He, 2000). As a valuable precedent for applying this approach in Haiti, ^{137}Cs measurements had already been successfully used in the neighbouring Dominican Republic to estimate soil erosion rates within the Nizao catchment (Nagle et al., 2000).

In January 2013 a sampling program was undertaken with the purpose of using the ^{137}Cs technique to document soil redistribution rates on a typical hills lope cultivated by local farmers in the traditional manner involving manual hand tools. The specific objectives of this investigation were: i) to determine the ^{137}Cs inventory and its soil vertical distribution at a suitable reference site in the explored region. ii) to use the ^{137}Cs measurements to estimate soil redistribution rates across a typical hillslope farming area.

2. Materials and methods

2.1. Study area description

The "Forêts des Pins" (18°19′41.63″ N; 71°47′12.16″ W) is a protected area established in 1937 by the Haitian authorities (Fig. 1). The entire region covers around 30000 ha. It is dominated by pine and mixed forest but also includes some areas with intense farming activities (< 10%).

This is a humid moist forest with altitudes ranging from 1500 to 2630 m a.s.l., featuring rugged slopes above 15%. The annual precipitation is in the range 1600–2000 mm, and the mean annual temperature is about 14 °C. July, August and September, the summer period, are the months of greatest precipitation, receiving between 200 and 300 mm per month when the temperature reaches maximum values between 15 and 22 °C. In February and March the rains are less abundant, with a monthly average of around 70 mm. The minimum temperature is usually recorded in January and February, with values lower than 5 °C. Temperatures below freezing are often observed (UNEP, 2013).

In this region of the Hispaniola Island, soils reflect the interaction between the tropical climate and topography as well as the volcanic, metamorphic or calcareous nature of the underlying parent material.

Weathering of the bedrock produces a fertile black surface horizon with a rather coarse granulometry and neutral pH. This horizon overlies a red horizon, clayey and infertile with a pH of 5.5–6.5. Leptosols and Calcisols are the prevailing soil formations. In the plains and valleys, soils are deep, clay-rich and often calcareous given rise to Vertisols and Luvisols. Under tropical forests, intense weathering results in highly leached Ferrasols. Weakly developed Cambisols provide the surface soils of many mountainous areas susceptible to erosion (European Union, 2015). The natural vegetation is typical of humid mountain forest dominated by *Pinus Occidentalis,* an endemic specie, which grows in high mountains and plateau regions. Areas belonging to the Forêts des Pins were originally exploited in 1945 by the "Société Haitiano-Américaine de Développement Agricole" (SHADA), but since the end of the 1940s, it has not suffered from any further anthropogenic disturbance. In contiguous areas and as a consequence of the population increase, traditional agriculture, with limited mechanization is undertaken by local farmers on hillside plots. Crop production is characterized by poor agricultural management practices and a lack of soil conservation measures. Food crops such as, maize (*Zea mays ssp.*), beans (*Phaseolus vulgaris L*), cassava (*Manihot esculenta*), potatoes (*Solanum tuberosum*), and cabbage (*Brassica oleracea*) occupy the areas of cleared forest (Dolisca et al., 2007).

2.2. Experimental design

2.2.1. Soil sampling at the reference site

Determining the ^{137}Cs inventory at a reliable reference site is a keyrequirement for using this radioisotope to document soil redistribution rates. The site should provide a reliable estimate of the reference inventory reflecting the original ^{137}Cs fallout, which is commonly characterized by an exponential ^{137}Cs depth distribution typical of an undisturbed location. A flat area of about 2 ha (18° 19′ 1.38″ N; 71° 46′ 30.58″ W) was selected as a potential reference site. In this area, the trees had been extracted by the SHADA Society more than 60 years ago and it has remained undisturbed since that time. To avoid boundary effects, a rectangular sampling grid was established in the centre of this area. This grid provided 9 sampling points in a 3 × 3 configuration, with a 12 m × 15 m spacing of the sampling points (see Fig. 2).

At each sampling point, samples were collected using a 5 cm diameter metal core tube. This was inserted into the soil using a manual hammer and removed with a steel lifting jack, which includes a lever and chain system to facilitate the extraction process. At each of the three diagonal points (P1, P5 and P9), 8 soil cores were collected within an area of 4 m^2, with care being taken to avoid any disturbance of the coring points during this exercise. All the coring points were at least 20 m from the nearest tree to avoid potential ^{137}Cs fallout interception by the canopy. The soil cores were extracted from the core tube in the field and sectioned into 5 cm depth increments, from the surface to a depth of 40 cm. At the remaining sampling points 5 soil cores were collected, at the vertices and in the centre of a 1.5 × 1.5 m square. After extraction from the core tubes, these cores were sectioned into10 cm depth increments from the surface to a depth of 40 cm. At each sampling point, the sections representing a given depth increment were combined to obtain a single composite sample from that depth for subsequent analysis.

2.2.2. Soil sampling in the cultivated field

The study site (18° 20′ 15.61″ N; 71° 49′ 12.72″ W) was located on a hillside, and was selected to be representative of an area where local families still used traditional manual farming methods mainly involving small hand hoes (see photo of a typical farmer and a panoramic image of the study site in Fig. 1). Given the tools used, the tillage activity affects only the superficial soil layers. Neither mechanized tools nor draft animal powered tillage were used at this site. Agricultural productivity is very low because the land is degraded. Plots are usually small and there are no modern agricultural inputs. Most of farms in the zone are owned by families practicing subsistence agriculture (UNEP, 2013). In the study area cultivation began at the beginning of 1992. Cultivation is generally carried out twice a year: during the months of February–March and in July–August. In order to investigate within-field variability of the ^{137}Cs inventories, three ca. 100 m downslope transects were established parallel to the slope (see Fig. 2). The distance between the transects was around 30 m. Along each transect, four representative sampling points were established. These were selected to avoid local topographic anomalies and were ca. 35 m apart. The values of slope gradient between the individual sampling points on the three transects are indicated on Fig. 2 and evidence significant variations along each transect and between points of the same transect. Samples were collected following the same experimental approach adopted at the reference site. At one point on each transect (P1) 8 soil cores were collected within an area of 4 m^2, avoiding any disturbance of the coring points prior to core collection. These cores were sectioned into 5 cm depth increments from the surface to a depth of 40 cm. The 8 sections representing an individual depth increment were bulked. At the other sampling points, 4 cores were collected, at the vertices of a 1.5 m × 1.5 m square, and these cores were sectioned into 10 cm depth increments from the surface which were bulked. On the lower part of the slope – where deposition was expected – longer cores were collected and these were sectioned to a depth of 50 cm.

2.3. Laboratory analysis

After classical pre-treatment (drying and sieving at 2 mm), measurements of the ^{137}Cs activity of all bulked depth incremental soil samples from the reference (n = 48) and the transect sites (n = 65) were undertaken at the FAO/IAEA Soil and Water Management & Crop Nutrition Laboratory at Seibersdorf, Austria. A high resolution HPGe coaxial detector mounted in a 100 mm lead shield, with a relative efficiency at 1.33 MeV of 115% was used to carry-out the γ-spectrometry analysis. Calibration of equipment, analysis and quality control of the measurements were performed following IAEA standard procedure (Shakhashiro and Mabit, 2009). The counting time for all samples was set at 50000s. The average analytical precision for the measurements was 10–15% at the 95% confidence level with a minimum detection activity (MDA) of 0.3 Bq kg^{-1}.

3. Results and discussion

3.1. The ^{137}Cs inventory and its depth distribution at the reference site

The measurements of ^{137}Cs activity concentration obtained for the bulked depth incremental samples collected from the reference site and the associated inventory values are presented in Table 1. Considering the nine sampling points at the reference site, the ^{137}Cs mean inventory is 600 Bq m^{-2}, with a standard deviation (*SD*) of 160 Bq m^{-2} and a coefficient of variation (*CV*) of 27%. However, sampling points P4 and P7, both located on the same side of the rectangular grid, exhibit a ca. 50% depletion of ^{137}Cs content in the upper soil layer when compared with the other sampled points (see Table 1). These two points have been treated as outliers. The reduced inventories at these two sampling points could reflect bioturbation or other causes of disturbance of the soil around the sampling point over the past ca. 50 years or possibly the past presence of a tree which increased the spatial variability of fallout receipt. A mean ^{137}Cs inventory based on the remaining 7 sites has been

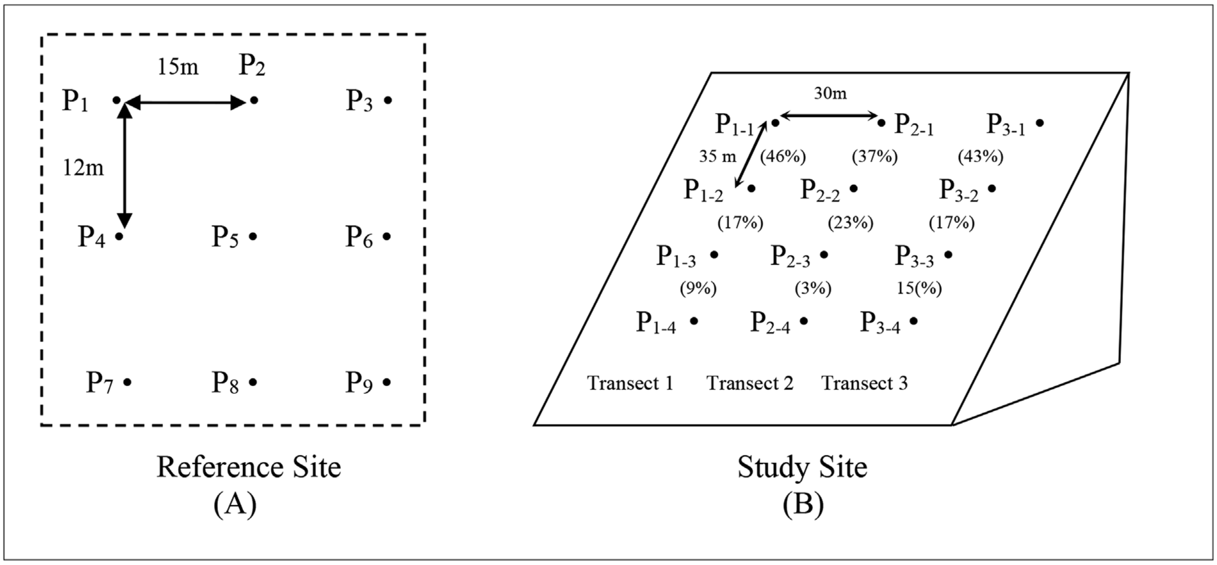

Fig. 2. Sampling design at the reference site (A) and at the agricultural study site (B).
NB: for the study site, between the sampling points of each transect, the slope (%) is given in parentheses.

calculated. This provides a value for the mean ^{137}Cs inventory of 670Bq m^{-2} with a SD of 100Bq m^{-2} and a CV of 15% (n = 7). Using these updated values and the approach proposed by Mabit et al. (2012), the estimated ^{137}Cs baseline inventory of 670 Bq m^{-2} has an acceptable allowable error (AE) of 12% at the 90% confidence level. This value for the reference inventory is in close agreement with the estimate of the contemporary inventory at this location equivalent to the expected ^{137}Cs fallout receipt (i.e. 570–630 Bq m^{-2}), based on its mean annual precipitation and its location relative to past nuclear weapons testing sites (Walling and He, 2000).

Table 1 provides information on the depth distribution of the ^{137}Cs inventory at the sampling points within the reference site, by listing the ^{137}Cs activity associated with the individual depth increments. These data indicate that most of the ^{137}Cs activity is found within the upper 10 cm of the soil profile and that activities decline rapidly below this level. An exponential depth distribution of this form is typical of an undisturbed soil profile experiencing neither erosion nor deposition (Walling and Quine, 1995). A representative ^{137}Cs depth distribution for the reference site has been derived by calculating the mean values of activity concentration associated with the 5 cm depth increments of the samples collected from the three diagonal sampling points (P_1, P_5 and P_9) (see Fig. 3).

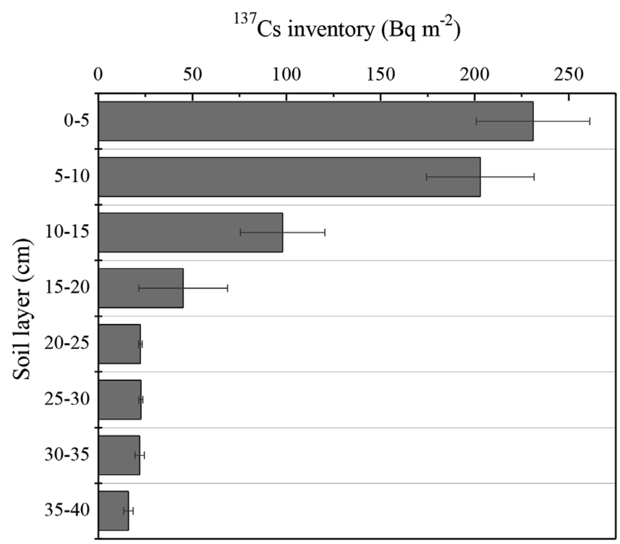

Fig. 3. ^{137}Cs inventory vs. soil depth at the reference site. Horizontal bars represent the ^{137}Cs inventory for the respective soil layer. For each bar the measurement uncertainty is indicated.

Table 1
^{137}Cs activity concentration and inventory at the reference site.

^{137}Cs activity concentration (Bq kg^{-1})										
Depth cm	P2	P3	P4	P6	P7	P8	Depth cm	P1	P5	P9
00–10	7.6 (0.7)	7.2 (0.7)	3.8 (0.6)	6.0 (0.6)	4.2 (0.5)	5.2 (0.6)	00–05	7.9 (0.7)	6.1 (0.6)	7.4 (0.6)
10–20	1.6 (0.5)	2.3 (0.5)	0.5 (0.0)	1.8 (0.5)	0.5 (0.0)	1.9 (0.5)	05–10	4.9 (0.6)	4.8 (0.6)	6.4 (0.9)
20–30	0.8 (0.5)	0.5 (0.0)	0.5 (0.0)	0.5 (0.0)	0.5 (0.0)	0.5 (0.0)	10–15	2.5 (0.5)	2.2 (0.5)	3.6 (0.5)
30–40	0.6 (0.0)	0.5 (0.0)	0.5 (0.0)	0.5 (0.0)	0.5 (0.0)	0.5 (0.0)	15–20	1.2 (0.6)	1.0 (0.5)	0.4 (0.0)
							20–25	0.5 (0.0)	0.5 (0.0)	0.5 (0.0)
							25–30	0.5 (0.0)	0.5 (0.0)	0.5 (0.0)
							30–40	0.6 (0.0)	0.5 (0.0)	1.8 (0.5)
Inventory Bq m^{-2}	670	690	340	680	400	530		610	670	870

In parentheses are reported the uncertainty associated with gamma spectrometry determined at 2σ.

Table 2
^{137}Cs activity concentrations and inventories for the sampling points within the study site.

^{137}Cs activity concentration (Bq kg^{-1})												
Depth cm	Transect 1				Transect 2				Transect 3			
	P1	P2	P3	P4	P1	P2	P3	P4	P1	P2	P3	P4
00–05	0.9 (0.5)				2.3 (0.6)				0.4 (0.0)			
05–10	0.7 (0.3)				0.3 (0.0)				1.8 (0.7)			
00–10		2.8 (0.7)	5.5 (0.8)	7.5 (0.7)		2.5 (0.5)	3.6 (0.7)	6.2 (0.8)		3.5 (1.4)	10.5 (1.1)	2.6 (0.6)
10–15	0.4 (0.0)				0.4 (0.0)				1.6 (0.5)			
15–20	0.3 (0.0)				0.4 (0.0)				0.3 (0.0)			
10–20		0.3 (0.0)	5.2 (0.7)	8.3 (0.8)		1.4 (0.5)	1.4 (0.4)	2.3 (0.7)		0.4 (0.0)	0.4 (0.0)	6.7 (0.9)
20–25	0.9 (0.5)				0.3 (0.0)				0.4 (0.0)			
25–30	0.4 (0.0)				0.3 (0.0)				0.4 (0.0)			
20–30		0.4 (0.0)	1.0 (0.5)	8.4 (0.7)		1.1 (0.5)	0.3 (0.0)	0.4 (0.0)		0.3 (0.0)	0.4 (0.0)	3.1 (0.5)
30–35	0.4 (0.0)				0.3 (0.0)				0.3 (0.0)			
35–40	0.8 (0.4)				0.3 (0.0)				0.3 (0.0)			
30–40		0.3 (0.0)	0.4 (0.0)	2.7 (0.5)		0.3 (0.0)	0.3 (0.0)	0.3 (0.0)		04 (0.0)	0.4 (0.0)	1.3 (0.6)
40–50				1.8 (0.5)				0.4 (0.0)				
Inventory Bq m^{-2}	200	340	960	2200	190	400	500	800	200	420	740	1200

In parentheses are reported the uncertainty associated with gamma spectrometry determined at 2σ.

3.2. ^{137}Cs inventories and activities at the study site

The values of ^{137}Cs activity concentration and ^{137}Cs inventory associated with the samples collected from the study site are presented in Table 2. The values of ^{137}Cs inventory range from 190 Bq m^{-2} to 2200 Bq m^{-2}, and thus deviate considerably from the reference inventory (670 Bq m^{-2}), indicating that substantial erosion and deposition has occurred within the site. For the 12 points sampled, seven are characterized by inventories less than the reference inventory and therefore erosion and five by inventories greater than the reference inventory and therefore deposition. Although the inventories documented along the three transects vary in relative magnitude, they follow a general pattern. This is characterized by a low inventory (indicating maximum erosion) near the top of the transect and inventories increasing downslope. The highest inventory values, which exceed the reference inventory (therefore indicating deposition), are found at sampling point 4 near the base of the transects, and for two of the three transects, an inventory value in excess of the reference inventory was also found at sampling point 3.

The data presented in Table 2 indicate that the ^{137}Cs content of the depth increments obtained from the sectioned soil cores varied significantly along each transect. For the surface soil layer (i.e. 0–10 cm), the ^{137}Cs activity concentration ranged from 0.8 to 7 Bq kg^{-1}for transect 1, from 1.6 to 6.2 Bq kg^{-1}for transect 2 and from 1.1 to 10.5 Bq kg^{-1} for transect 3. In general, the highest ^{137}Cs activities are found in the uppermost depth increment and activities increase downslope towards the base of the transects.

Fig. 4 provides a summary of the general shapes of the ^{137}Cs depth distributions and their variation according to position on the transect. For all sampling points, the ^{137}C content of the surface layers (0–10 cm; 10–20 cm) was significantly higher than found in the deeper soil layers. At sampling point P4 (lower part of sampled area) on transects 1 and 3, the ^{137}Cs content is comparable in the first two soil layers (i.e. 0–10 cm, 10–20 cm).

As noted above, the occurrence of the lowest ^{137}Cs inventories near the top of the slope indicates that these points are associated with the highest erosion rates, despite the high slope gradients between this sampling point and sampling point 2 on all three transects. Because of these high gradients, erosion rates might be expected to increase downslope due to the increasing slope length, which is likely to be coupled with an increasing runoff depth and therefore transport capacity. The absence of such a trend and the occurrence of the lowest inventories (and highest erosion rates) at the top of the slope could suggest that tillage translocation represents an important cause of soil redistribution, in addition to water erosion which will reflect effects of rain splash in detaching soil particles and the role of surface runoff in transporting the detached sediment and mobilising further sediment. However, the limited depth of cultivation and soil translocation associated with manual hoeing means that tillage effects are likely to be limited. Nevertheless, at the top of the slope the limited supply of sediment from upslope by tillage translocation could result in increased erosion rates. As a result, the soil redistribution rates can be expected to reflect changes in slope gradient which will influence the intensity of both tillage translocation and water erosion. The low gradients at the foot of the slope are marked by deposition, due to a reduction in the transport capacity of runoff and the reduced effectiveness of tillage translocation, resulting in the upslope supply exceeding the downslope export and therefore deposition.

The data presented in Table 2 and Fig. 4 indicate that the ^{137}Cs depth distributions found along the transects are characterized by significant variability in their general shape. In general, however, the sampling point on the middle part of the slope (i.e. points P2 and P3 of each transect) which could be expected to be characterized by soil loss, are characterized by high values of ^{137}Cs activity near the surface. This situation is atypical and unexpected for a cultivated site, where tillage is expected to result in a near-uniform ^{137}Cs activity concentration throughout the plough depth. The apparent lack of vertical mixing shown by the ^{137}Cs depth distributions is seen to reflect the particular way in which the local farmers work the land and the use of manual cultivation practices. This results in very limited mixing of the individual depth increments. Sampling point $P_{2\text{-}3}$ has a ^{137}Cs inventory similar to that found at the reference site, which is indicative of a low rate of soil loss. The information on the ^{137}Cs depth distribution available for this sampling point (see Table 2 and Fig. 4) indicates that this is very similar to that at the reference site, suggesting that it reflects the same downward diffusion and migration processes as those operating at the reference site and that the impact of tillage mixing within a shallow surface layer is essentially the same as the bioturbation that affects the surface horizon of the uncultivated reference site (see Fig. 3).

3.3. Estimating soil redistribution rates on the cultivated slope

Use of ^{137}Cs measurements to estimate soil redistrubution rates involves the use of a range of different mathematical approaches - termed

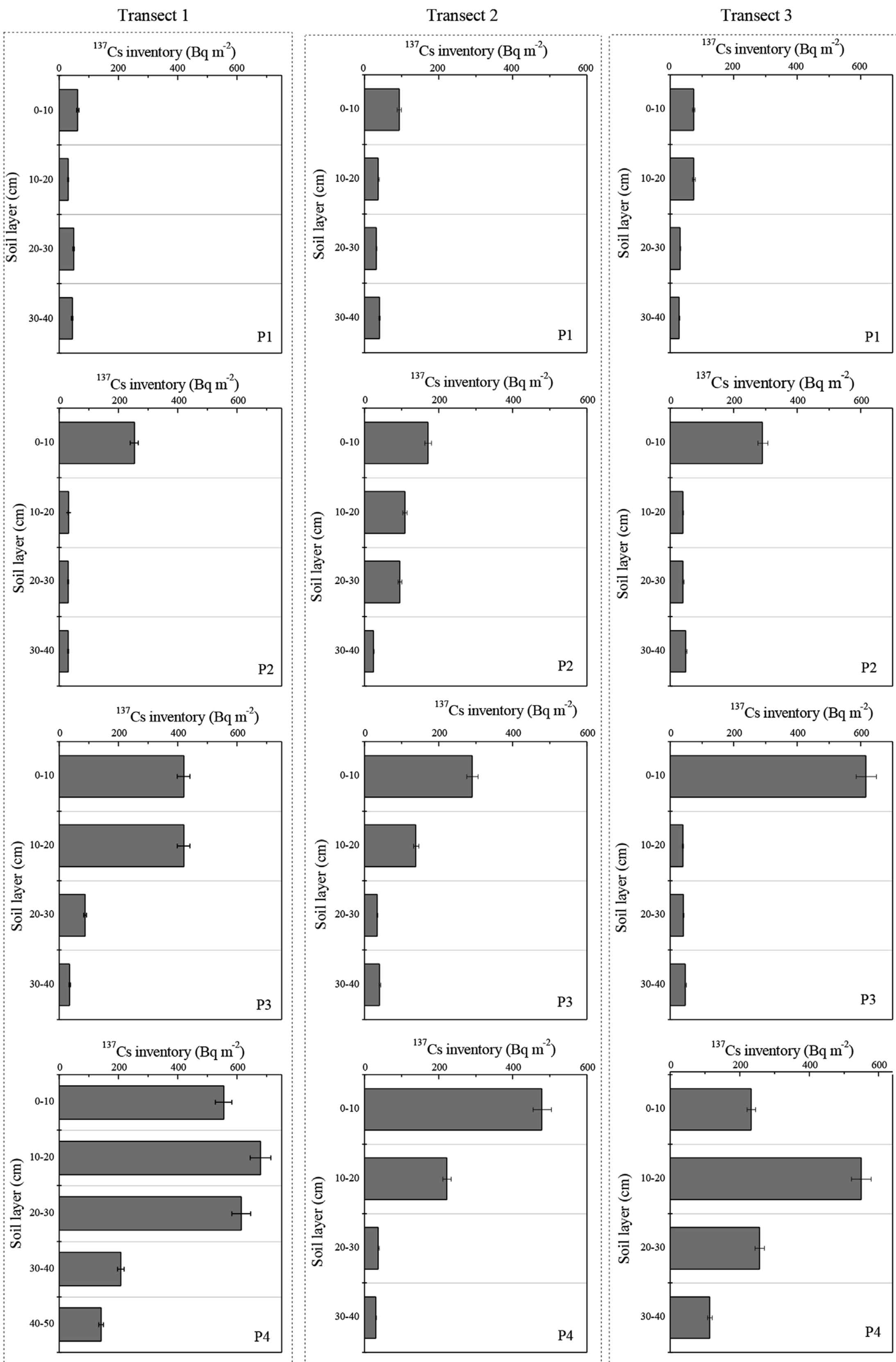

Fig. 4. ^{137}Cs inventory vs. soil depth at the study site. For the three transects and for each sampling points (P1, P2, P3 and P4) the horizontal bars represent the ^{137}Cs inventory for the respective soil layer. For each bar the measurement uncertainty is shown.

conversion models - to convert the ^{137}Cs measurements to quantitative estimates of soil erosion and deposition rates (see Walling et al., 2002, 2014; Zapata, 2002). Such approaches are based on the assumption that a sampling point with a total radionuclide inventory less than the local reference inventory represents an eroding site, whereas a point with a total radionuclide inventory greater than the local reference inventory is assumed to be a depositional site. The models make it possible to estimate the erosion or deposition rate by comparing the inventory measured at a sampling point with the reference inventory. A key factor in the choice of an appropriate conversion model is the distinction between cultivated and uncultivated soils. In the first case, the conversion model needs to take account of the effects of tillage in mixing the radionuclide within the plough layer and incorporating new soil from beneath the plough depth into the plough layer as the surface is lowered by erosion. In the case of uncultivated soils, the available models focus on removal of soil from the surface and use information on the form of the ^{137}Cs depth distribution to establish a relationship relationship between surface lowering and reduction of the ^{137}Cs inventory of the soil. In most situations, the depth distribution will reflect the exponential form associated with uncultivated soils. Initiallyit was assumed that because the study site was cultivated, estimates of soil redustribution rates would be derived using one of the mass balance models for cultivated soils (e.g. mass balance model 2) described by Walling et al. (2002) which is now available as part of a simple to apply Excel-based software package covering several different conversion models for both cultivated and uncultivated soils (see Walling et al., 2011, 2014). After the ^{137}Cs measurements on the samples collected had been completed, it rapidly became apparent that this approach could not be justified. The information on the depth distribution of ^{137}Cs activity within the soils of the study area presented above did not conform to the fundamental assumptions of the available mass balance models. These models assume that the plough layer is well mixed by tillage and is characterized by essentially uniform ^{137}Cs activity throughoutits depth, that all, or at least most, of the ^{137}Cs inventory is found within the plough layer and that the plough layer extends to a depth similar to, or greater than, the depth of the base of the ^{137}Cs depth distribution found at the reference site. Fig. 4 clearly indicates that this is not the case. As indicated above, the ^{137}Cs depth distribution shown for sampling point $P_{2\text{-}3}$, which has a^{137}Cs inventory close to that of the reference inventory, is very similar to that of the reference site (see Fig. 3). This similarity of the depth distribution suggests that in the absense of erosion, the evolution of the ^{137}Cs depth distribution in undisturbed soil is closely similar to that associated with the ^{137}Cs vertical transport in soil that has been subject to shallow cultivation.

As a first step in the search of an appropriate conversion model suitable for the particular conditions of the study site, we propose a mathematical approach for describing the vertical migration of ^{137}Cs in the soil of a reference site. In previous investigations, where the ^{137}Cs distribution patterns are comparable to those found in these Haitian soils, the diffusion-convection equation (*DCE*) has been successfully applied (e.g. He and Walling, 1997; Toso and Velasco, 2001; Bossew and Kirchner, 2004). This approach assumes that the ^{137}Cs prevailing transport processes are molecular diffusion, hydrodynamic dispersion and convection. In brief, this model proposes the following equation to describe the ^{137}Cs vertical distribution in a soil profile (He and Walling, 1997):

$$\frac{\partial C(z,t)}{\partial t} = D_e \frac{\partial^2 C(z,t)}{\partial z^2} - v_e \frac{\partial C(z,t)}{\partial z} - \lambda C(z,t) \tag{1}$$

where z (kg m^{-2}) is the cumulative mass depth in the soil; t (years) is time (if it is assumed that the ^{137}Cs fallout input represents weapons testing fallout, t = 0 is usually assumed to be 1963 or 1964, the time of maximum fallout in the northern and southern hemispheres respectively); $C(z,t)$ (Bq kg^{-1}) is the ^{137}Cs activity concentration, D_e (kg^2 m^{-4} y^{-1}) is the effective diffusion coefficient (which comprises molecular diffusion and hydrodynamic dispersion), v_e (kg m^{-2} y^{-1}) is the effective downward migration rate, and λ (y^{-1}) is the ^{137}Cs decay constant.

Bossew and Kirchner (2004) reported D_e and v_e values of more than 500 soil profiles and for different radionuclides. For the velocity v_e, the values ranged from 0.1 to 0.5 cm y^{-1} (equivalent to 0.6–7 kg m^{-2} y^{-1}), following a normal distribution. For some profiles, the maximum concentration was at the ground surface implying that $v_e \approx 0$. While D_e values ranged from 0.05 to 0.5 cm^2 y^{-1} (close to 3–70 kg^2 m^{-4} y^{-1}) and followed a lognormal distribution.

The following equation is frequently used as a first approximation of the solution of Eq. (1) (Konshin, 1992):

$$C(z,t) = \frac{C_0}{2\sqrt{\pi D_e t}} e^{-\lambda t} e^{-(z-v_e t)^2 / 4D_e t} \tag{2}$$

where C_0 (Bq m^{-2}) is the initial ^{137}Cs inventory at the time of fallout.

Approximate values of the effective diffusion coefficient (D_e) and the migration rate (v_e) can be obtained using the following equations (Walling et al., 2002):

$$D_e \approx \frac{(N_p - W_p)^2}{t - 1963} \tag{3}$$

$$v_e \approx \frac{W_p}{t - 1963} \tag{4}$$

where t is the sampling year, W_p(kg m^{-2}) is the mass depth of the maximum ^{137}Cs concentration and N_P (kg m^{-2}) is the distance between the mass depth of the maximum ^{137}Cs concentration and the mass depth where the ^{137}Cs concentration reduces to 1/e of the maximum value. The year 1964 should be used for sites in the southern hemisphere.

Using Eqs. (3) and (4), the depth distribution of ^{137}Cs activity concentration obtained at the reference site (Fig. 3) is characterized by D_e = 37 kg^2 m^{-4} y^{-1} and v_e = 0.33 kg m^{-2} y^{-1}. When combined with Eq. (2) they provide a good fit to the measured depth distribution at the reference site (Fig. 5A) and can be used to model the evolution of the ^{137}Cs depth distribution at the study site where tillage does not seem to substantially affect the mechanisms of ^{137}Cs migration within the soil profile.

Considering the spatial and temporal evolution of ^{137}Cs in the soil of the study site, it is now necessary to select the suitable conversion model to convert differences between the ^{137}Cs inventory documented for the sampling points within the cultivated study site and established reference inventory into erosion or sedimentation rates. The most commonly used conversion models for undisturbed rangeland or pasture soils are the profile distribution model and the diffusion and migration (D&M) model (see Walling and He, 1999; Walling et al., 2002). The former can be expected to overestimate erosion rates, since the exponential ^{137}Cs depth distribution documented at the reference site at the time of sampling is assumed to have existed at the study site since the main period of ^{137}Cs fallout (i.e. 1963). This does not correspond to the situation under consideration since the orginal fallout would originally have been found close to the surface and the considerably deeper depth distribution found at the time of sampling would have evolved by downward diffusion and migration over the subsequent decades. To obtain more reliable estimates of soil redistribution rates, there is a need to take account of the temporal evolution of the ^{137}Cs depth distribution. The D&M model sets out to do this and models the evolution of the depth distribution in response to the fallout inputs, subsequent downward diffusion and migration and radioactive decay. This conversion model was seen as providing the best approach for use in the Haiti study. However, in the existing software (Walling et al., 2011, 2014) the model assumes that soil redistribution was ongoing prior to the late 1950s which marked the beginning of the main period of ^{137}Cs fallout. In the study area, land clearance and cultivation were known to have occurred in 1992 and available local knowledge indicated that little or no erosion occurred at the study site prior to 1992 and that significant soil redistribution commenced in 1992. The

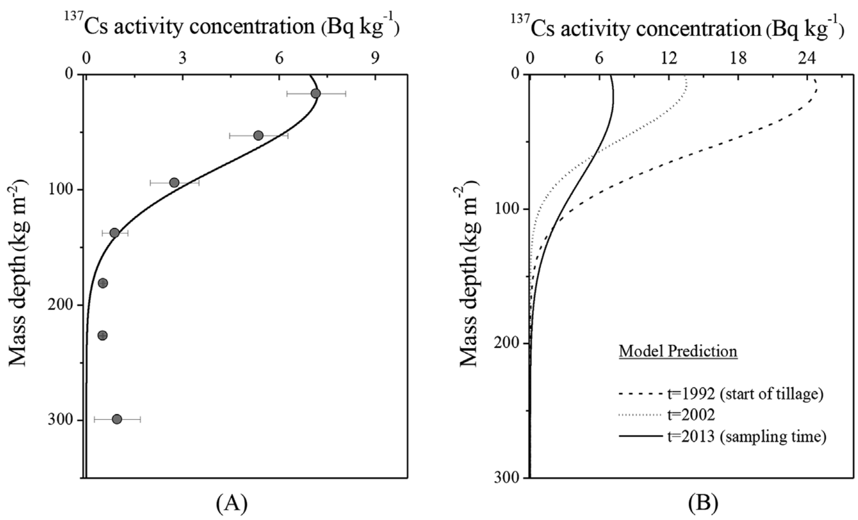

Fig. 5. (A) ^{137}Cs activity concentration vs. mass depth for the reference site documented in 2013, showing the measurement uncertainty associated with the individual depth increments and the fitted diffusion-convection model curve. (B) Predicted ^{137}Cs specific activity vs. mass depth for the years: 1992, 2002 and 2013.

increased and decreased ^{137}Cs inventories relative to the reference inventory documented at the study site were therefore assumed to reflect erosion and deposition occurring during the period extending from 1992 to the time of sample collection in 2013.

In summary, the proposed modification of the D&M conversion model assumes that: *a.* the soil redistribution processes began simultaneously with the start of tillage in 1992 and continued until the sampling date (2013), and *b.* the initial condition of the ^{137}Cs depth distribution at the study site when farming activities started is given by the DCE equation for t = 1992. This is the same depth distribution associated with the reference site at this point in time, since both sites were uncultivated and experienced no soil redistribution prior to 1992.

Figure 5B presents ^{137}Cs depth distribution at the reference site for t = 1992 (when farming activities began at the study site), for t = 2002 and for the sampling time (t = 2013).

Adapting the *D&M* conversion model (Walling and He, 1999, Walling et al. 2002) for the sampling points where the ^{137}Cs inventory is less than the reference site, the erosion rates (R) have been calculated using the following equation:

$$\sum_{t=1992}^{2013} RC_s(t)\exp(-\lambda(2013-t)) = I_{SP} - I_{RS} \tag{5}$$

where:

R: erosion rate (kg m^{-2} y^{-1})
C_s: mean ^{137}Cs activity concentration in the eroded soil layer (Bq kg^{-1})
t: time in years, the summation index, with lower limit 1992 and higher 2013.
I_{RS}: ^{137}Cs inventory at reference site in 2013 (Bq m^{-2})
I_{SP}: ^{137}Cs inventory at a sampling point in the study site (Bq m^{-2})

Where the measured ^{137}Cs inventory exceeds the reference value, the sampling point is assumed to be located in an area of deposition. The increase in inventory is assumed to be the result of the accumulation of asoil layer of thickness R' per year throughout the whole tillage period. For a deposition point on a transect, the mean ^{137}Cs specific activity of the sediment eroded from upslope will reflect both the surface activity of the upslope eroding sampling points on the transect and the erosion rate calculated for each of those points. Thus, for a given year, the ^{137}Cs activity concentration of the deposited soil layer should reflect both the ^{137}Cs activity concentration of the soil eroded from the upslope area and the erosion rate representing the amount of soil mobilised from different parts of this upslope area. If we assume continuity down each transect across the study site and that the upslope area is represented by the upslope sampling points, the activity of the eroded soil can be calculated as the erosion rate weighted mean activity of the surface soil at those upslope points in a given year. If there are n eroding sampling points upslope of the depositional area, where $C_i(t)$ (i = 1, 2, …n) represents, respectively, the ^{137}Cs mean activity concentration of the eroding sediment at time t, and Ri is the erosion rate of each sampling eroded point, then the weighted mean ^{137}Cs activity concentration of the deposited surface layer at time t is determined as:

$$C_s^{wf}(t) = \frac{R_1}{\sum_i R_i}C_1(t) + \frac{R_2}{\sum_i R_i}C_2(t) + \cdots\cdots + \frac{R_n}{\sum_i R_i}C_n(t) \tag{6}$$

Considering the entire period of tillage, from Eq. (6) and taking into account the difference between the ^{137}Cs inventory of a sampling point characterized by deposition and the reference inventory, the mean deposition rate at this sampling point, R', can be estimated using the following equation:

$$\sum_{t=1992}^{2013} R'C_s^{wf}(t)\exp(-\lambda(2013-t)) = I_{SP} - I_{RS} \tag{7}$$

where:

R': deposition rate (kg m^{-2} y^{-1})

C_s^{wf}: weighted mean ^{137}Cs activity concentration of the deposited surface layer taking account of the erosion rate at the upslope sampling points (Bq kg^{-1})

The estimates of erosion and deposition rate (i.e. the R and R' values) obtained for the sampling points within the study site are reported in Table 3. This table also includes the weighting factors determined from Eq. (6). and then used in Eq. (7) for calculating the weighted mean ^{137}Cs concentration activity. Fig. 6 provides information on the soil redistribution rates and the elevations of the individual sampling points along the transects across the study site. The results indicate that soil loss is greatest at the top of the slope (P1) and is similar for each transect (between −22 and −23 t ha^{-1}y^{-1}). Deposition occurs at the base of the slopes, but the deposition rates show high variability between the transects. The highest deposition rate (71 t ha y^{-1}) is found at point P4 on transect 1. At the bottom of transects 2 and transect 3 deposition rates are respectively 6.1 t ha y^{-1} and 23 t ha y^{-1}. Because of the reconnaissance nature of the study reported, the number of samples collected from the three transects was limited. This precludes reliable estimation of rates of net soil loss from the slopes of the study site, since it is not possible to define accurately the zones along the transects experiencing soil loss or deposition, in order to calculate the gross soil loss and the amount of soil redeposited further down the

Table 3
Erosion rate (*R*) and deposition rate (*R'*) for each sampling point.

	Sampling Point	Inventory Bq m^{-2}	Weighting factors	R t ha y^{-1}	R' t ha y^{-1}
Transect 1	P1-1	200		−23	
	P1-2	340		−15	
	P1-3	960	0.60 (T1P1) + 0.40		14
	P1-4	2200	(T1P2)		71
Transect 2	P2-1	190		−23	
	P2-2	400		−13	
	P2-3	500		−7.7	
	P2-4	800	0.53 (T2P1) + 0.29 (T2P2) + 0.18 (T2P3)		6.1
Transect 3	P3-1	200		−22	
	P3-2	420		−12	
	P3-3	740	0.66 (T3P1) + 0.34		3.4
	P3-4	1200	(T3P2)		23

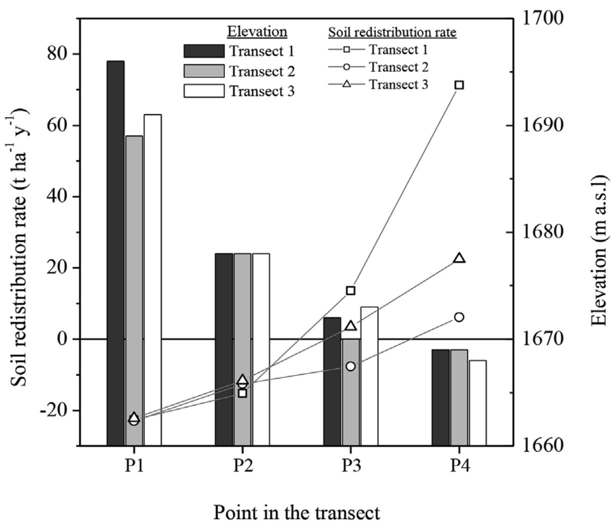

Fig. 6. Soil redistribution rate and elevation for each sampling point in the three transects.

slope. However, the results presented in Fig. 6 suggest that deposition represents a significant component of the sediment budget of the slopes and that rates of net soil loss are significantly less that rates of gross soil loss. This suggests that the efficiency of sediment delivery from the slopes to adjacent watercourses is not high.

The values of erosion rate obtained in the present investigation are in line with the findings of the Food and Agriculture Organization of the United Nations (FAO, 2015) which concludes that for hilly croplands in tropical and subtropical areas, erosion may reach values up to 50–100 t ha^{-1} y^{-1}, with average rates of 10–20 t ha^{-1} y^{-1}. This erosion level is associated with evidence of removal of surface horizons, indicating that the original biotic functions are partly destroyed (FAO, 2015).

4. Conclusion

This study reports the first use of the ^{137}Cs technique in Haiti. Attention focussed on quantifying the magnitude of soil redistribution rates within a cultivated area located on a steep slope in a cleared zone within an area of tropical Haitian forest known as the "Forêts des Pins". The reference inventory measured within an adjacent undisturbed area was 670 ± 100 Bq m^{-2} (CV = 15%; AE = 12%; n = 7). Inventories measured within the study site ranged from 200 to 2200 Bq m^{-2}. Problems were encountered in deriving estimates of soil redistribution rates from the ^{137}Cs measurements using standard procedures, since the ^{137}Cs depth distributions documented in the cultivated study area showed no evidence of significant tillage mixing. This was attributed to the traditional manual tillage practices which involved limited disturbance of the surface soil. This situation meant that it was not possible to apply standard conversion models developed for use in areas with cultivated soils. The Diffusion and Migration model, originally developed for undisturbed (i.e. pasture or rangeland soils) was modified for use in the study area. Estimates of soil redistribution rates obtained from the measured inventories using the modified conversion model developed for the study site ranged for erosion rates from −7.7 to −23 t ha^{-1} y^{-1} and for deposition rates from 3.4 to 71 t ha^{-1} y^{-1}at the base of the slopes. The estimates soil redistribution rates obtained confirm that soil loss represents a serious environmental problem in Haiti, and the need to adopt conservation measures to ensure the sustainability of Haitian agroecosystems. Future studies could usefully employ a more detailed sampling programme to provide information on the sediment budget of the study slopes and more particularly the relative magnitude of rates of gross and net soil loss and thus the efficiency of sediment delivery connectivity between the slopes and adjacent watercourses. The study is seen as confirming the potential for using ^{137}Cs in Haiti to provide information on rates of soil loss and soil redistribution. The use other FRNs such as ^{7}Be and/or $^{210}Pb_{ex}$ in the "Forêts des Pins" could provide additional information about the soil erosion/sedimentation processes in the study area and could in the future prove useful in evaluating the effectiveness of conservation practices in controlling and mitigating the degradation of Haitian soils.

In a more general context, the study is seen as demonstrating the need to evaluate carefully the results of ^{137}Cs measurements obtained from a study site, in order to confirm that the basic assumptions for their use in estimating soil redistribution rates using standard procedures are met. In this study, the ^{137}Cs depth distributions documented for the cultivated study site did not conform to expectations. It was necessary to adapt an existing conversion model developed for uncultivated soils. Although existing procedures (e.g. Zapata, 2002) will be applicable in most studies, the use of the ^{137}Cs approach in an ever-increasing range of environments means that situations where such standard procedures are inappropriate will inevitably arise. Users should be prepared to critically assess the data collected and, where necessary, adapt standard procedures.

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