**BRAZILIAN ARCHIVES OF
BIOLOGY AND TECHNOLOGY**

A N   I N T E R N A T I O N A L   J O U R N A L

Assesment of Soil Erosion by ^{137}Cs Technique in Native Forests in Londrina City, Parana, Brazil

Avacir Casanova Andrello[1]*, Carlos Roberto Appoloni[1] and Virgílio Franco do Nascimento Filho[2]

[1]*Departmento de Física; CCE; Universidade Estadual de Londrina; C. P.: 6001, 86051-390; Londrina - PR - Brasil.* [2]*Loboratóerio de Instrumentação Nuclear; Centro de Energia Nuclear em Agricultura; Universidade de São Paulo; Av. Centenário, 303; C. P.: 96; 13400-970; Piracicaba - SP - Brasil*

ABSTRACT

The aim of this work was to assess the soil erosion process in native forest by the ^{137}Cs methodology. The mass balance model was applied to assess the rates of soil loss in three native forests around of Londrina city, Paraná, Brazil. ^{137}Cs distribution depth was of exponential type for the three forests and ^{137}Cs inventory was 241 Bq m^{-2} for Mata 1, 338 Bq m^{-2} for Mata 2 and 325 Bq m^{-2} for Mata UEL. The soil loss value calculated for three native forests was: 6,684 kg ha^{-1} yr^{-1} for Mata 1, 1,788 kg ha^{-1} yr^{-1} for Mata 2 and 4,524 kg ha^{-1} yr^{-1} for Mata UEL.

Key words: Cesium-137, mass balance model, native forest

INTRODUCTION

Soil erosion is the foremost cause of the ecosystem degradation. Many studies have been carried out to understand this problem, but they are time consuming and expensive. Moreover, these studies were conducted in the plots that did not represent major areas adequately being applied only for the cultivated soils or undisturbed soils of pasture. In undisturbed soil forest, the hydro erosion impact depends on the type and quantity of canopy and very few studies have been dedicated to the qualitative and quantitative understanding of this. In this ecosystem, the crash drop of the rain is reduced by treetop and mold on soil surface, so that hydro erosion is reduced. However, soil redistribution in native forests can occur by other processes, such as diffusion, migration of soil particles in the soil matrix, and bioturbation and

self-evolution of forests. For these reasons, the conventional methods are limited to quantify soil redistribution, and consequently, the soil erosion rates for this environment. A conventional method is the RUSLE (Revised Universal Soil Loss Equation; Renard et al, 1991) that has been utilized in study of soil erosion for the cultivated soils or pastures, but very few were applied to undisturbed soil forests (Özhan et al, 2005; Erskine et al, 2002; Wallbrink et al, 1998).
Another method that has been applied to study the soil erosion and deposition is the ^{137}Cs methodology. ^{137}Cs is an anthropogenic radionuclide (half life of 30.2 years) introduced in the environment by fallout of nuclear weapons tests and Chernobyl accident (Andrello et al, 2003). When it is deposited on the soil surface, it is rapidly and tightly adsorbed by the micaceous minerals (Tamura and Jacobs, 1960; Sawhney,

---

* Author for correspondence

1971; Poinssot et al, 1999, Onodera et al, 1998) and its redistribution in the environment occurs by the physical processes (principally by erosion). Organic matter quantity in the superficial soil horizons determines the ^{137}Cs availability for the downward diffusion and migration in the soil matrix. Many papers have shown that when ^{137}Cs is adsorbed by the minerals parts of the soil, its desorption by the natural chemical processes does not exist or is very unready (Vlacke and Cremers, 1994; Facchinelli et al, 2001). Hence, ^{137}Cs has become a soil marker and has been utilized in the soil erosion assessment (Andrello et al, 2003 and 2004). Several empirical and theoretical models have been proposed to soil erosion quantification (losses or gains) by ^{137}Cs redistribution (Walling and He, 1999). To use these models, a reference inventory value for ^{137}Cs is determined and then compared with ^{137}Cs inventory determined for the sampled soil. If studied soil has less ^{137}Cs than reference value, then the soil loss would occur, otherwise, soil deposition would take place. Thus, the reference ^{137}Cs inventory is necessary to establish a correlation between the ^{137}Cs redistribution and soil erosion. ^{137}Cs depth distribution in undisturbed soil can be used to show the bioturbation occurring in the soil since ^{137}Cs deposition by fallout. In recent years, many papers have been devoted to study the migration of ^{137}Cs in the soil system, but, this process is not fully understood because it's complex and a large number of parameters may affect the migration of ^{137}Cs in the soil system. It is well known that bioturbation can contribute significantly to the vertical transport of fallout radionuclides that might have a considerable impact on the long-term in the depth distribution profile of radionuclides.

Bioturbation can occur due to a variety of endopedonic animals' activity, of that the earthworms are very active in consuming and excreting large amounts of the soil. It can transport the soil components downward and upwards in the soil matrix, which can change considerably the morphological structure of the soil. Although in the grassland soils some information on the effect of bioturbation on the transport of fallout radionuclides is available, but this information is not available for the forest soils.

The aim of this work was to study the soil erosion occurring in three forests around Londrina city, Parana, Brazil and to assess the long-term bioturbation in the soil profile of these forests by ^{137}Cs depth distribution.

MATERIALS AND METHODS

The Model

The mass balance model developed by Kachanoski and de Jong (1984) and adapted by Yang et al. (1998) was utilized in this work. This model considers the ^{137}Cs annual deposition through overall fallout period (1954 to early 1980s) and can be applied to determine the soil erosion rates in undisturbed soils. For any year t, the mass balance model can be represented as:

$$S_t = S_{t-1} + F_t - E_t \qquad (1)$$

where S_t is the total ^{137}Cs inventory in the soil profile at year t (Bq m^{-2}), S_{t-1} is the total ^{137}Cs inventory in soil profile at year t-1 (Bq m^{-2}), F_t is the total ^{137}Cs deposited by fallout at year t (Bq m^{-2}), E_t is the amount of ^{137}Cs lost of soil profile at year t (Bq m^{-2}), where t changes from 1 to N (N = M – 1954, M is the sampling year). Assuming that there is a constant erosion rate, this model can be utilized to assess the total ^{137}Cs content in the soil at end of elapsed time. Although, assuming that the soil erosion rate is constant does not correspond to the true, it can be applied because an average soil loss will be obtained for the studied time period.

If the total amount (C_t) and annual fraction (r_t) of ^{137}Cs deposited by fallout on the region during an elapsed time period are known, F_t can be expressed as:

$$F_t = r_t\, C_t \qquad (2)$$

Due to the lack of total ^{137}Cs deposition data of monitoring stations for the most regions of the world, the reference ^{137}Cs inventory (C_R, in Bq m^{-2}) determined in the study area in the specific year t can be used instead of C_t, so that F_t can be rewrite as:

$$F_t = r_t\, C_R \qquad (3)$$

Although the reference ^{137}Cs inventory is easily determined in the study area, the same not occur to the annual fraction (r_t). However, despite of total ^{137}Cs level deposited is different in several regions of the South Hemisphere, it may be admitted that the annual fraction (r_t) of the ^{137}Cs deposited is the same for all these regions. Based on data of

Walling and He (1997), values of r_t for South Hemisphere were determined and was shown in Table 2, in the form $R_t = 100 \times r_t$. The contribution due to Chernobyl accident was not considered in these values.

Study Area
In this work, the mass balance model was applied to assess the soil loss in the three native forests around Londrina city, Paraná State, Brazil. The native forests were labeled as Mata 1, Mata 2 and Mata UEL. The vegetation of these native forests is lowland rainforest and the soil belongs to the group Oxisols by U. S. soil taxonomy. The slope of Mata 1 changes from 10 to 20%, while slope of Mata 2 is lower than 1% and the slope of Mata UEL is around of 5%. Mata 1 had a 3 cm thickness of mulch on soil surface and a great quantity of little roots down to 20 cm depth. Mata UEL had approximately 2 cm thickness of mulch on soil

surface and little roots down to 5 cm depth. Mata 2 had a thickness of mulch minor than 1 cm on soil surface and almost nothing of little roots. The granulometric properties of the soil for three native forests are shown in Table 1. These values were obtained by homogenization of three cores of each native forest resulting in one sample that was analyzed to granulometric properties. This procedure was utilized because the soil of the three native forests was the same group. The studied areas are located around the coordinates of $23^0 16'$ S and $51^0 17'$ W. The mean elevation of the areas is around 665 m above sea level. The regional climate is classified as humid sub-tropical (mean temperature = 20.7 ^oC) with rainfall throughout the year, with a great probability of winter dryness (Corrêa et al., 1982). The average annual rainfall was 1615 mm yr^{-1}, based on measurements carried out at surrounding meteorological sites, from 1975 to 1998.

Table 1 - Granulometric properties of the soil of the three native forests.

Depth (cm)	Sand g/kg	Silt g/kg	Clay g/kg	pH	CEC (mE/100g)	O.C. g/kg
0-20	120	190	690	6.8	14.6	22.5
20-30	120	170	710	6.5	12.8	16
30-80	90	110	800	6.4	8.9	7.6

O.C. – Organic Carbon

Sampling and Analysis
In the three native forests the sampling point was chosen the farthest possible of big trees and probable holes left for dead trees, in order to avoid the possible ^{137}Cs accumulation due to the rain water flowed by the tree stem and root during the fallout period. In each native forest a point was sampled 1, 2, 4 and 5 cm increment at depth down to 25 cm. In order to diminish the punctual variability, each sample, with 2 kg of soil, was obtained from three replicates sampled from 1 m^2 area. The samples were air dried for 48 h, grounded for < 2 mm, and packaged in Marinelli beaker of 2 l for ^{137}Cs activity determination.

The activity determination was realized by gamma-ray spectrometry with 10 and 20% relative efficiency HPGe detector. The detection time was 216,000 s for the 10% detector and 54,000s for the 20% detector. The software MAESTRO model A65-B32, version 4.10, was utilized for the spectra analysis.

RESULTS

^{137}Cs distribution profile for three native forests was of exponential type (Fig. 1). This was in accordance with other results for the undisturbed soils (Walling and Quine, 1995). A mathematical model utilized to describe this distribution form could be given by the equation:

$$C_s = a\,e^{-bz} \qquad (a > 0 \text{ and } b > 0) \qquad (4)$$

where C_s is ^{137}Cs concentration (Bq kg^{-1}) at the depth z (m) and a and b are the function coefficients. The a coefficient represented the amount of ^{137}Cs that should be there in the superficial soil if soil loss or soil redistribution did not occur. The b coefficient represents the shape of the ^{137}Cs depth distribution profile. Hence, greater b showed more shallow ^{137}Cs depth distribution profiles.
Following Yang et al (1998), if ^{137}Cs distribution profile can be represented by a function f(z),

where $f(z)$ is a function of depth z (m), then ^{137}Cs concentration C_s (Bq kg^{-1}) is:

$$C_s = f(z) \tag{5}$$

Hence, the reference ^{137}Cs inventory (C_R, in Bq m^{-2}) can be represented as:

$$C_R = \int_0^H D f(z) \, dz \tag{6}$$

where D is the soil bulk density (kg m^{-3}) and H is depth (in m) in which ^{137}Cs still can be detected.

Let h be the mean annual thickness of the superficial soil loss, the mean annual ^{137}Cs loss (λ), relative to the total ^{137}Cs present in the soil profile, can be determined by the expression:

$$\lambda = \frac{\int_0^h D f(z) \, dz}{C_R} = \frac{\int_0^H D f(z) \, dz - \int_h^H D f(z) \, dz}{\int_0^H D f(z) \, dz} \tag{7}$$

in which λ may assume values between 0 and 1. The ^{137}Cs loss in year (t) can be determined by:

$$\lambda_t = \frac{E_t}{S_{t-1} + F_t} \qquad (0 \leq \lambda \geq 1) \tag{8}$$

where E_t (Bq m^{-2}) is the ^{137}Cs lost at year t, S_{t-1} (Bq m^{-2}) is total ^{137}Cs inventory at year t-1 and F_t (Bq m^{-2}) is the amount of ^{137}Cs deposited by fallout at year t. In undisturbed and stable soils, it is expected that ^{137}Cs distribution profile would be similar to several years, i.e., remain being exponential type. Rogowski and Tamura (1970) and Filipovic-Vicenkovic et al (1991) have observed this behavior during several years for the undisturbed soils. If there is no major perturbation in the soil, an exponential ^{137}Cs distribution depth may stay year after year, hence, in first approximation it can be assumed:

$$\lambda_t = \lambda = \text{constant} \ (t = 1, 2, \ldots, N; 0 \leq \lambda \geq 1) \tag{9}$$

Using equation (9) in (8), we get:

$$E_t = \lambda (S_{t-1} + F_t) \tag{10}$$

Equations (10) and (3), equation (1) can be re-written as follows:

$$S_t = (1 - \lambda)(S_{t-1} + r_t C_R) \ (t = 1, 2, \ldots, N) \tag{11}$$

Solving this equation until t = N, it would be:

$$S_N = (1 - \lambda)^N r_1 C_R + (1 - \lambda)^{N-1} r_2 C_R + \\ + (1 - \lambda)^{N-2} r_3 C_R + \ldots + (1 - \lambda) r_N C_R \tag{12}$$

where S_N is ^{137}Cs inventory in eroded soil profile at sampling year.

Let ^{137}Cs inventory at sampling year be denoted by C_E, then $S_N = C_E$. Keeping C_R in equation (12), it would be:

$$1 - \frac{C_E}{C_R} = 1 - \{(1 - \lambda)^N r_1 + (1 - \lambda)^{N-1} r_2 + \\ + \ldots + (1 - \lambda) r_N\} \tag{13}$$

Let

$$Y = \left(\frac{C_R - C_E}{C_R}\right) \times 100 \tag{14}$$

and

$$R_t = r_t \times 100 \tag{15}$$

where Y is percentage of ^{137}Cs loss in relation to the reference inventory at sampling year and R_t is the percentage fraction of total ^{137}Cs fallout deposition at year t.

Table 2 showed that $R_t = 0$ after 1983, then equation (13) became, (for M > 1983):

$$Y = 100 - \{(1 - \lambda)^{28} R_1 + (1 - \lambda)^{27} R_2 + \\ + (1 - \lambda)^{26} R_3 + \ldots + R_{29}\}(1 - \lambda)^{M-1983} \tag{16}$$

As the values for R_t are given in Table 2 and the values for C_R can be calculated by equation (6) and (4), then, it is only necessary to determine the values of C_E to assess the mean annual ^{137}Cs loss (λ) for each native forest. The values of C_E for each native forest was determined using the following equation:

$$C_E = \sum_{i=1}^z C_i \times D_i \times I_i \tag{17}$$

where C_i is the ^{137}Cs activity in the i th depth (Bq kg^{-1}), D_i is the soil bulk density in the i th depth

(kg m^{-3}), I$_i$ is the depth increment in the i th depth (m) and z is the maximum number of depth increment until the depth where ^{137}Cs can be detected. The values of C$_E$ for three native forests are presented in Table 3.

As the function f(z) that represents the ^{137}Cs distribution profile was known for th three native forests, it is possible to determine the reference inventory C$_R$ for each native forest through the use of equations (4), (5) and (6) and the values for *a* and *b*. Then, it was possible to determine Y by equation (14) for each native forest. The values of C$_R$ and Y for each native forest are shown in Table 3. As possible to λ assume values from 0 to 1, it was realized several iterations with equation (16), with λ changing from 10^{-4} to 0.4 and M changing from 1988 to 2030. Graphics for Y, with Y changing from 0% to 100%, was obtained from these iterations. Utilizing these graphics, values of λ is possible to be determined, since values of Y are known, and vice-versa. With the values of Y obtained for the three native forests and with the

graphics given in Fig. 2, the λ value was calculated for each native forest, and the mean annual thickness of soil loss (h) for each native forest was determined by equation (7). The λ value and h for each native forest are shown in Table 4. Knowing h, it was possible to calculate the mean annual soil loss by the following equation:

$$E_R = h \times D \times 10000 \qquad (18)$$

where E$_R$ is the mean annual soil loss (kg ha^{-1} yr^{-1}), h is the mean annual thickness of soil loss (m), D is the soil bulk density (kg m^{-3}) and 10000 is a conversion factor from hectare to square meter.

A mean soil bulk density of 1100 kg m^{-3} was utilized to calculate the mean annual soil loss for the three native forests. The values for E$_R$ are shown in Table 4.

Table 2 – Values of ^{137}Cs annual deposition (F$_t$) and the ^{137}Cs annual fraction (R$_t$) deposition to the South Hemisphere (based on Walling and Quine, 1997).

Year (t)	F$_t$ (Bq m^{-2})^a	R$_t$ (%)^b	Year (t)	F$_t$ (Bq m^{-2})^a	R$_t$ (%)^b
1954 (1)	9.10	1.3	1970 (17)	26.29	2.6
1955 (2)	27.92	3.9	1971 (18)	32.12	3.1
1956 (3)	28.57	3.9	1972 (19)	23.48	2.2
1957 (4)	32.48	4.4	1973 (20)	19.22	1.7
1958 (5)	36.56	4.8	1974 (21)	29.50	2.6
1959 (6)	30.61	3.9	1975 (22)	20.12	1.7
1960 (7)	24.36	3.1	1976 (23)	15.45	1.3
1961 (8)	46.30	5.7	1977 (24)	10.54	0.9
1962 (9)	69.25	8.3	1978 (25)	10.78	0.9
1963 (10)	74.59	8.7	1979 (26)	8.28	0.7
1964 (11)	83.96	9.6	1980 (27)	5.65	0.4
1965 (12)	89.92	10	1981 (28)	0	0
1966 (13)	39.96	4.4	1982 (29)	0	0
1967 (14)	28.63	3.1	1983 (30)	6.05	0.4
1968 (15)	20.92	2.2	>1983 (M-1983)	0	0
1969 (16)	42.82	4.4	Total	893.29	100

^{a}Corrected values to 2001.
^{b}R$_t$ = 100 x r$_t$

Table 3 – Values of *a* and *b* coefficients, total ^{137}Cs inventory (C$_E$), reference inventory (C$_R$) and percent ^{137}Cs loss (Y) obtained for three native forest.

|	a (Bq kg^{-1})	b (m^{-1})	C$_E$ (Bq m^{-2})	C$_R$ (Bq m^{-2})	Y (%)
Mata 1	3.18	10.26	240.6	296.6	18.83
Mata 2	5.09	14.48	337.6	365.4	7.61
Mata UEL	3.31	9.25	325.3	369.5	11.96

Table 4 – Values of mean annual ^{137}Cs loss (λ), mean annual thickness of the superficial soil loss (h) and the mean annual soil loss $(E)_R$ for three native forest.

	λ	h ($\times 10^{-2}$ m)	E_R (kg ha^{-1} yr^{-1})
Mata 1	0.0057	0.056	6684
Mata 2	0.0022	0.015	1788
Mata UEL	0.0035	0.038	4524

(a) Mata 1
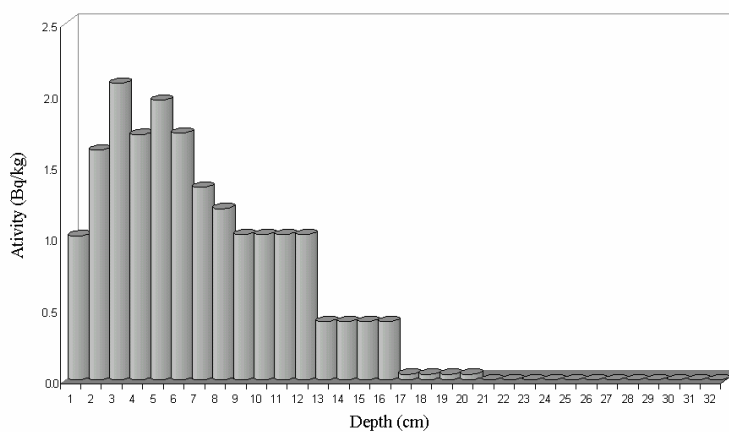
(b) Mata 2
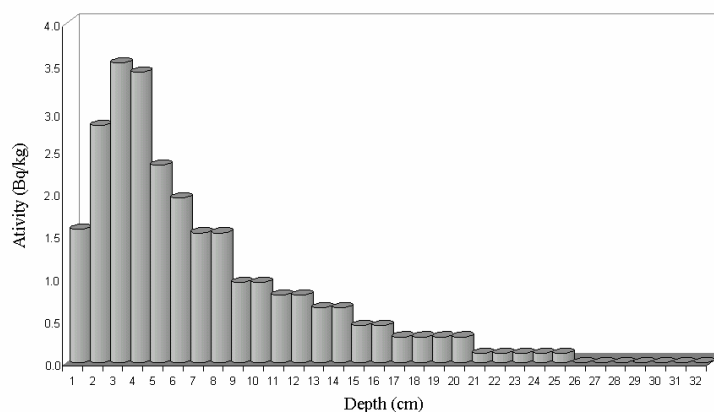
(c) Mata UEL
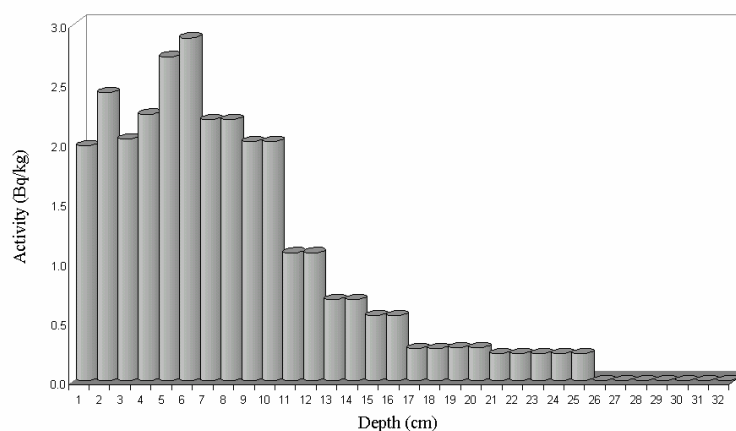
Figure 1 – ^{137}Cs distribution profile in relation to depth for three native forests.

DISCUSSION

The ^{137}Cs distribution profile for three native forests is presented in Fig. 1. Only Mata 2 presented the ^{137}Cs profile of the exponential type. Mata 1 and Mata UEL did not have present a good exponential type because in these forests, there was a great quantity of the little roots. In Mata 1, these little roots did not allow to sample in increments depth of 2 cm below 8 cm depth, which probably generated a poor exponential fit. Moreover, Fig. 1a showed that the ^{137}Cs distribution down to 6 cm was approximately constant, except in 0-1 cm depth increment. This uniformity of ^{137}Cs in depth from 1 to 6 cm could have been generated by a bioturbation, as during the sampling a high quantity of the native tree seeds at 0-6 cm depth was observed, which only could have occurred due to a soil mixing. An important transport mechanism for the radionuclides in the soil, which might have a considerable impact on the long-term predictions, was the bioturbation (Bunzl, 2002). Fig. 1.c presented the ^{137}Cs profile for Mata UEL that showed a good exponential type below 5 cm depth, although the ^{137}Cs distribution down to 4 cm was approximately uniform. This uniformity down to 4 cm could be the consequence of the little roots and beetles activity in this increment depth (0-4 cm). The Mata 1 and Mata UEL had great quantity of mulch, 3 cm and 2 cm, respectively, that was generated by the high annual litter fall, which gave a good condition to pedivores activity, especially the earthworms. Because Mata 2 had a little mulch, 1 cm approximately, the pedivores activity practically did not exist, hence, the ^{137}Cs distribution depth showed a good exponential type, similar to the results presented by Walling and Quine (1995) for the undisturbed soils.

Using the equation (4), the coefficients *a* and *b* was obtained by nonlinear curve fitting and these coefficients described the ^{137}Cs distribution depth for each native forest. Table 3 present the values of *a* and *b* coefficients for each native forest. The difference between these values was due to the lack of similarity in the ^{137}Cs distribution profile for three native forest and by poor exponential fit for Mata 1 and Mata UEL, as discussed above. In spite of this difference, it could be observed (Table 3) that only the C_R value for Mata 1 differed from the others, which occurred too for the C_E value.

Table 3 showed that Mata 1, Mata 2 and Mata UEL had ^{137}Cs loss (Y) of 19, 8 and 12%, respectively, in relation to the total ^{137}Cs deposited by the fallout in the period of 47 years (from 1954 to 2001). Using these values for Y, the constant erosion rates (λ) were determined through Fig. 2, which resulted in the values of 0.0057, 0.0022 and 0.0035 for Mata 1, Mata 2 and Mata UEL, respectively.

The mean annual thickness of the superficial soil loss (h) was determined employing the values of λ for the three native forests. The values of h were 0.056, 0.015 and 0.038 cm for Mata 1, Mata 2 and Mata UEL, respectively (Table 4). Multiplying the values of h for each native forest by the elapsed time period (47 years) from the beginning of fallout (1954) to the sampling date (2001), the total thickness of superficial soil loss was obtained. The values of h in the sampling points were 2.6 for Mata 1, 0.7 for Mata 2 and 1.8 cm for Mata UEL. Fig. 1-a showed that Mata 1 presented a ^{137}Cs deficiency in the 0-2 cm profile, which was concordant with the value of soil loss of 2.6 cm obtained for this point. In this point, the ^{137}Cs activity determined for the 0-2 cm profile was due to the ^{137}Cs dilution in the adjacent downward layers by the bioturbation (Tyler et al, 2001). The bioturbation in this point was down to 6 cm and the ^{137}Cs concentration was approximately uniform too, which indicated that ^{137}Cs dilution had occurred in this profile (0-6 cm). Thus, the rate of annual soil loss calculated for Mata 1 could be major.

Assuming that the minor inventory value C_E determined for Mata 1 was due to a high soil loss, and that the lack of exponential fit for ^{137}Cs distribution depth for this native forest did generate a misleading value of C_R, the value of 2.6 cm obtained for the total thickness of superficial soil loss for Mata 1 was underestimated.

Admitting that the value of C_R for Mata 1 was of the same order that the value of C_R for others two native forests, that is, C_R for Mata 1 was of order 360 Bq m^{-2}, the new value of mean annual thickness of superficial soil loss for Mata 1 was 0.15 cm, which represented a 7 cm of total soil loss in the period of 47 years.

Comparing the ^{137}Cs distribution profile of the three native forests presented in Fig. 1, it was observed that the ^{137}Cs activity was detected down to 25 cm for Mata 2 and Mata UEL, while for

Mata1 was detected only down to 20 cm. This difference of 5 cm between the ^{137}Cs distribution profile of the native forests showed that the value of 7 cm of soil loss for Mata 1 was not overestimated. If the value of 0,0056 cm of the mean annual thickness of soil loss for Mata 1 was considered, the mean annual soil loss would be 6684 kg ha^{-1} yr^{-1}, but considering the value of 0.15

cm, the value of mean annual soil loss would be 16500 kg ha^{-1} yr^{-1}. However, both the values of mean annual soil loss were high for the undisturbed soil in this native forest, where canopy coverage were almost 100% and the soil are very stable.
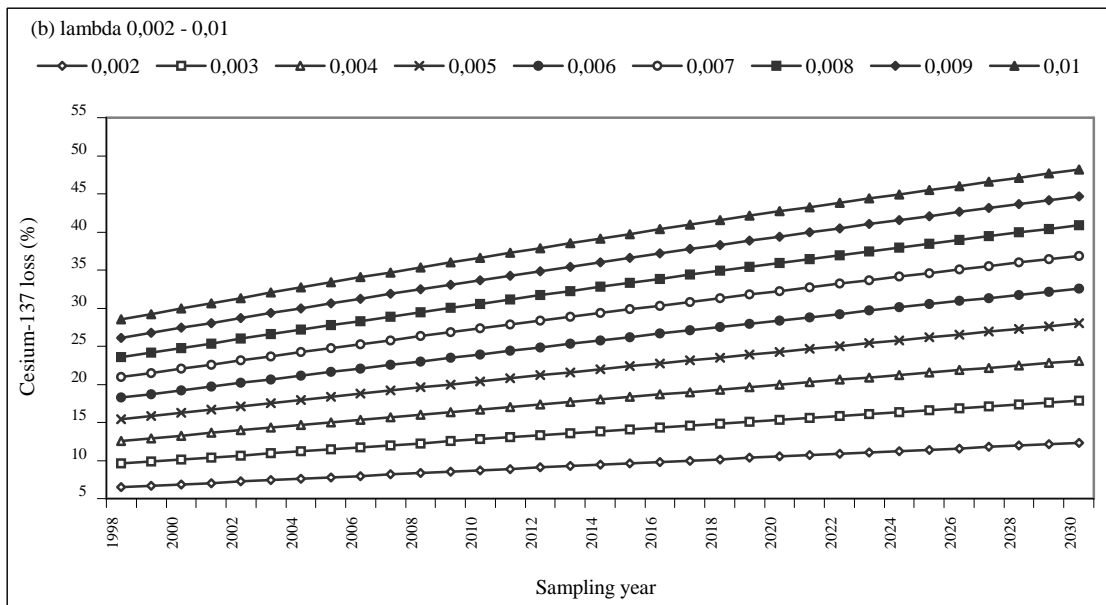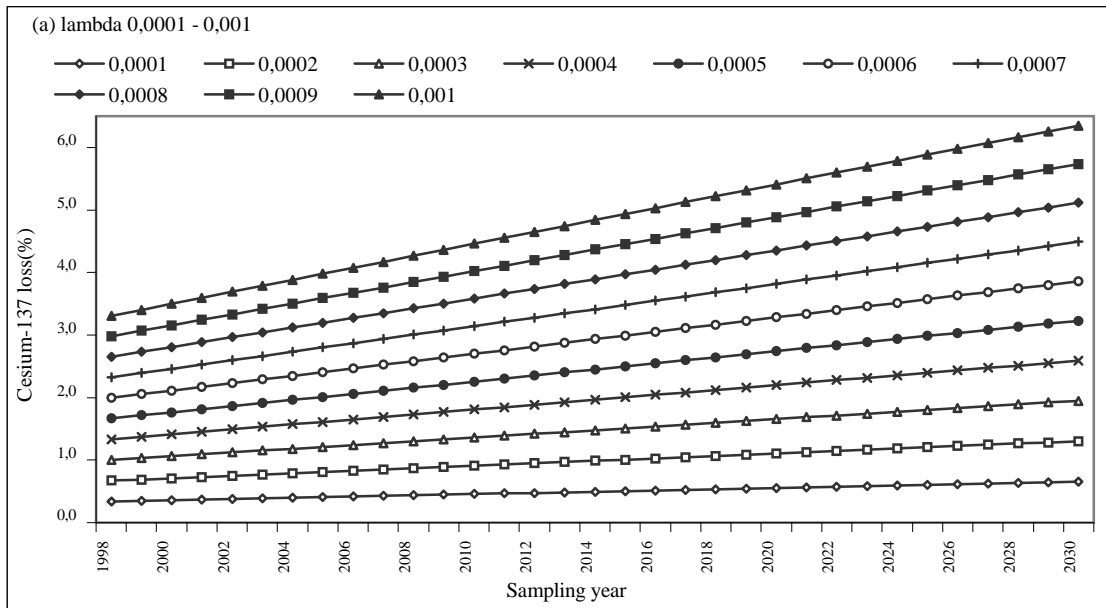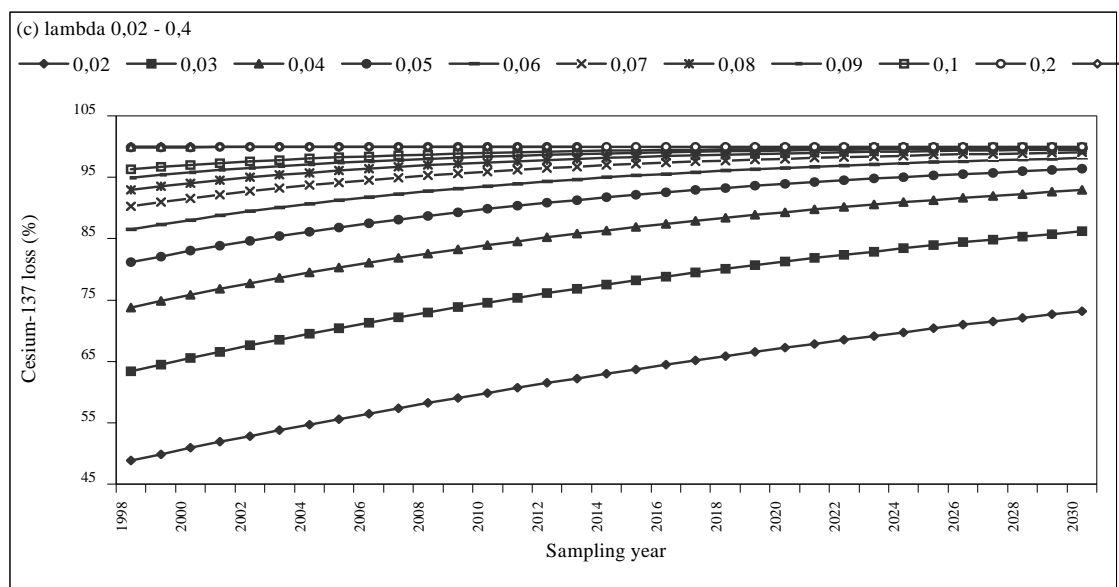
Figure 2 – Graphics obtained from interactions of equation (16) for percent ^{137}Cs loss (Y), for mean annual ^{137}Cs loss (λ) and for the sampling year (M)

Garcia-Oliva et al (1995) presented the values of soil loss in the forest that were close to the values obtained for Mata 1. The studied forest by Garcia-Oliva et al (1995) was situated in a hillslope area with slope of 40% that was twice greater than that of Mata 1. Although there was differences between the soils and slopes of Mata 1 and the studied forest by Garcia-Oliva et al (1995), the soil loss values in these two studies were in agreement and indicated that for undisturbed soils in native forest there were soil loss.

CONCLUSION

The mean annual soil loss for Mata 2 was 1788 kg ha^{-1} yr^{-1} and for Mata UEL 4524 kg ha^{-1} yr^{-1}. It would be interesting to compare the values of the soil loss for the native forests determined by the other methods with those obtained in this study, but due to the lack of these data, it was not possible. This comparison would be important to estimate the viability of the methodology employed in this work. A more rigorous study in these native forests could be realized by a grid sampling, in order to represent the overall area of the native forest and to determine the extension and actual rates of soil loss in these forests. Although the studied native forests were sampled

in one point bulking three replicates, it was possible to quantify that there was a soil loss and that more studies should be realized with the objective to understand and quantify the soil loss process in this kind of environments.

RESUMO

O processo de erosão de solo em floresta nativa tem sido pouco investigado. Como a metodologia do césio-137 dá resultados tanto de taxas de erosão de solo como a bioturbação no perfil de solo, ele tem sido usado para avaliar o processo de erosão de solo nestes ecossistemas. O modelo de balanço de massa foi aplicado para avaliar as taxas de perdas de solo em três florestas nativas na região de Londrina, Paraná, Brasil. A distribuição em profundidade do césio-137 para as três florestas é do tipo exponencial. O inventário de césio-137 foi de 241 Bq m^{-2} para Mata 1, 338 Bq m^{-2} para Mata 2 e 325 Bq m^{-2} para Mata UEL.O valor de perda de solo calculado para Mata 1 foi 6,684 kg ha^{-1} yr^{-1}, 1,788 kg ha^{-1} yr^{-1} para Mata 2 e 4,524 kg ha^{-1} yr^{-1} para Mata UEL.

REFERENCES

Andrello, A. C., Guimarães, M. F., Appoloni, C. R. Nascimento Filho, V. F. (2003). Use of cesium-137 methodology in the evaluation of superficial erosive processes. *Brazilian Archives of Biology and Technolog,* **46,** 307-314.

Andrello, A. C., Appoloni, C. R., Guimarães, M. F. (2004). Soil erosion determination in a watershed from nothern Parana (Brazil) using Cs-137. *Brazilian Archives of Biology and Technology,* **47,** 659-667.

Corrêa, A.R.; Godoy, H.; Bernardes, L.R.M. (1985), Características climáticas de Londrina. Londrina, Brazil.

Erskine, W. D., Mahmoudzadeh, A., Myers, C. (2002), Land use effects on sediments yields and soil loss rates in small basins of Triassic sandstone near Sydney, NSW, Australia. *Catena,* **49,** 217-287.

Facchinelli, A., Gallini, L., Barberis, E., Magnoni, M., Hursthouse, A. S., (2001). The influence of clay mineralogy on the mobility of radiocaesium in upland soils of NW Italy. *J Environ Radioact,* **56,** 299-307.

Filipovic-Vincekovic, N., Barisic, D., Masic, N., Lulic, S. (1991). Distribution of fallout radionuclides through soil surface layer. *J Radioanal Nucl Chem*, **148,** 53-62.

Garcia-Oliva, F., Martinez Lugo, R., Maass, J. M. (1995). Long-term net soil erosion as determined by ^{137}Cs redistribution in an undisturbed and perturbed tropical decidous forest ecosystem. *Geoderma*, **68,** 135-147.

Kachanoski, R. G., and De Jong, E. (1984). Predicting the temporal relationship between soil cesium-137 and erosion rate. *J Environ Qual,* **13,** 301-304.

Onodera, Y., Iwasaki, T., Ebina, T., Hayashi, H., Torii, K., Chatterjee, A., Mimura, H. (1998). Effect of layer charge on fixation of cesium ions in smectites. *J Contaminant Hydrol*, **35,** 131-140.

Özhan, S., Nihat Balci, A., Özyuvaci, N., Hizal, A., Gökbulak, F., Serengil, Y. (2005). Cover and management factors for the Universal Soil-Loss Equation for forest ecosystems in the Marmara region, Turkey. *Forest Ecology and Management*, **214**, 118-123.

Poinssot, C., Baeyens, B., Bradbury, M. H. (1999). Experimental and modeling of caesium sorption on illite. *Geochim Cosmoch Acta*, **63,** 3217-3227.

Renard, K. G., Foster, G. R., Weesies, G. A., Porter, J. P. (1991). Revised universal soil loss equation. *J Soil Water Conserv*, **46,** 30-33.

Rogowski, A. S., and Tamura, T. (1970). Erosional behavior of cesium-137. *Health Phys*, **18,** 467-477

Sawhney, B. L. (1972). Selective sorption and fixation of cations by clay minerals: a review. *Clay Clay Mineral,* **20,** 93-100.

Tamura, T., Jacobs, D. G. (1960). Structural implications in cesium sorption, *Health Phys*, **2,** 391-398.

Tyler, A. N., Carter, S., Davidson, D. A., Long, D. J., Tipping, R. (2001). The extent and significance of bioturbation on ^{137}Cs distributions in upland soils. *Catena*, **43,** 81-99.

Vlacke, E., Cremers, A. (1994). Sorption-desorption dynamics of radiocaesium in organic matter soils. *Sci Total Environ*, **157,** 275-283.

Yang, H., Chang, Q., Du, M., Minami, K., Hatta, T. (1998). Quantitative model of soil erosion rates using ^{137}Cs for uncultivated soil. **Soil Sci**, **163,** 248-257.

Wallbrink, P. J., Roddy, B. P., Olley, J. M. (1998). Using Cs-137 and Pb-210 to quantify soil erosion and deposition in forest. Paper presented at South Pacific Environmental Radioactivity Association Workshop, 16-20 Febuary, Christchurch, New Zealand.

Walling, D. E., and Quine, T. A. (1995). Use of fallout radionuclide measurements in soil erosion investigations. *In: Nuclear Techniques in soil-plant studies for Sustainable Agriculture and Environmental Preservation*, IAEA Publish, Vienna.

Walling, D. E., and He, Q. (1997). Models for converting ^{137}Cs measurements to estimates of soil redistribution rates on cultivated and uncultivated soils. *In A contribution to the IAEA coordinated research programmes on soil erosion (D1.50.05) and sedimentation (F3.10.01)*, IAEA Publish, Vienna, 27p.

Walling, D. E. and He, Q. (1999). Improved models for estimating soil erosion rates from cesium-137 measurements. *J Environ Qual*, **28,** 611-622.