

# CAESIUM-137 AS A SOIL EROSION TRACER: A REVIEW

GRZEGORZ J. PORĘBA

Department of Radioisotopes, Institute of Physics, Silesian University of Technology,  
Krzywoustego 2, 44-100 Gliwice, Poland  
(e-mail: Grzegorz.Poreba@polsl.pl)

**Key words:**

<sup>137</sup>Cs, SOIL EROSION,  
MATHEMATICAL  
MODELS

**Abstract:** This paper describes a method of soil erosion and soil movement measurements using <sup>137</sup>Cs as a tracer. This technique has been successfully employed for more than 30 years. Caesium-137 is a valuable tracer to study soil erosion. Its chemical and biological movement in soil is limited. Practically all redeposition of caesium-137 in soil after fallout occurs with soil particles due to strong fixation of caesium onto soil particles. This technique overcomes many problems of the traditional techniques to study soil erosion. First of all, this technique gives quantitative results of soil erosion and deposition for the medium-term (about 40 years). Moreover, these results are possible to obtain in a relatively short time compared to the traditional methods. The <sup>137</sup>Cs method to obtain quantitative value of soil erosion and deposition requires the availability of undisturbed sites to obtain the reference value of <sup>137</sup>Cs fallout for the study area and a numerical relationship. Currently, there are many calibration relationships available and the calculation of the soil erosion from <sup>137</sup>Cs activity data strongly depends on the chosen relationship and the parameters that were used for a particular model.

## 1. INTRODUCTION

Soil erosion is a serious environmental problem. Traditional methods used to measure soil erosion are time consuming and the results obtained for an experimental plot are usually incomparable with one another. The use of <sup>137</sup>Cs to measure soil erosion overcomes some of the limitations of the traditional methods (Stach, 1996a and 1996b). These methods are a valuable alternative compared to the traditional methods. <sup>137</sup>Cs measurements can be used as a basis for studying both the spatial variability of soil loss and its magnitude. This method can be used also for identifying sediment sources (Walling and Quine 1991; Wallbrink and Murray, 1993; Wallbrink *et al.*, 1999; Olley *et al.*, 1993). The first attempts to use <sup>137</sup>Cs measurements for estimation of soil erosion were carried out in the 1960-ties (Yamagata *et al.*, 1963; Rogowski and Tamura, 1965) and since then caesium-137 has been used to study soil erosion and deposition. Caesium <sup>137</sup>Cs is a

fission product, it is a gamma emitter with energy of 661.6 keV and its half-life is 30.02 years. These properties make the <sup>137</sup>Cs isotope a valuable tracer of soil erosion in the medium term. In addition, the counting of the sample is very easy. For the last 35 years, the fallout of <sup>137</sup>Cs has been widely used as an environmental tracer to study soil erosion (Ritchie and McHenry, 1990; Zapata, 2003). This paper describes a <sup>137</sup>Cs method for the determination of soil erosion as well as the evolution of this method. The basis of this method is presented as well as models to convert <sup>137</sup>Cs activity data into soil erosion values. The problems connected with using <sup>137</sup>Cs to study (measure) soil erosion are also discussed. The paper presents examples of application of this method to study soil erosion in Poland. Last review papers about using of <sup>137</sup>Cs for study soil erosion in Poland were published about ten years ago (Stach, 1996a and 1996b) and also Ritchie and McHenry (1990) published their review article more than ten years ago.

## 2. THE BASIS OF THE $^{137}\text{Cs}$ METHOD TO STUDY SOIL EROSION

The background of this method is well described in a number of papers (Stach, 1996b; Ritchie and McHenry, 1990). The method is based on the comparison of the  $^{137}\text{Cs}$  inventories for a sampling point with a reference inventory. The reference inventory presents the local input fall-out of  $^{137}\text{Cs}$ . This means that for a reference site neither erosion nor deposition of soil occurred. It should be mentioned that even for places where only global fallout of caesium was noticed, the spatial variability of the inventory could be relatively high and caution should be exercised (Sutherland, 1994 and 1996). The activity of  $^{137}\text{Cs}$  in soil samples could be measured by means of gamma spectrometry. The method to determine the activity of  $^{137}\text{Cs}$  and other naturally occurring gamma-ray emitters in sediments was described in detail in Ritchie and McHenry (1973a) and Murray *et al.* (1987). The inventory of  $^{137}\text{Cs}$  might be calculated for example according to Sutherland (1994). Generally, in a place where soil erosion occurs, the  $^{137}\text{Cs}$  inventory is smaller than the reference inventory. On the other hand, for a soil deposition area, the  $^{137}\text{Cs}$  inventory is greater than the reference inventory. This simple comparison allows recognizing erosion and deposition areas; however, to obtain quantitative estimation of soil erosion one of the relationships should be used (Walling and Quine, 1990; Walling and He, 1999). Those relationships allow converting  $^{137}\text{Cs}$  data into erosion rate data. The models are discussed below.

## 3. THE HISTORY OF USING $^{137}\text{Cs}$ AS A SOIL EROSION TRACER

The use of fallout radioisotopes to study soil erosion was started at the end of 1950s. The first fallout radioisotope which was used as a soil erosion and soil movement tracer, was strontium-90 (Menzel, 1960; Frere and Roberts, 1963). They concluded that the  $^{90}\text{Sr}$  loss from a field was due to soil erosion. Moreover, Frere and Roberts (1963) found no loss of  $^{90}\text{Sr}$  from grassland.

One of the first attempts to use caesium to study soil erosion and movement was presented by Yamagata *et al.* in the early 1960 (Yamagata *et al.*, 1963). In that paper, authors concluded that the loosing of caesium and strontium from catchments was induced by runoff. Rogowski and Tamura (1965) used a  $^{137}\text{Cs}$  technique for small experimental plots in Tennessee. The authors added  $^{137}\text{Cs}$  to the small experimental plot and they found an exponential relationship between the soil loss and  $^{137}\text{Cs}$  loss. This measurement was relatively short (81 days) and as a result no conclusion on long-term behaviour of  $^{137}\text{Cs}$  can be made. The same authors (Rogowski and Tamura, 1970a; 1970b) published the results of a two year study of  $^{137}\text{Cs}$  loss and soil loss. They confirmed that movement of caesium by runoff and erosion may be significant. They also found that  $^{137}\text{Cs}$  is strongly adsorbed onto soil particles and this adsorption is very rapid. Moreover, further movement of  $^{137}\text{Cs}$  in soil by infiltration is limited.

There is also a body of research devoted to the study of the chemical behaviour of caesium in soil and investigating the mechanism of caesium sorption onto soil particles (Tamura and Jacobs, 1960; Tamura, 1964; Schulz *et al.*, 1960; Schulz, 1965; Davis, 1963; Coleman *et al.*, 1963). The influence of the other ions on the caesium sorption was also investigated (Coleman *et al.*, 1963) as well as the various humic substances (Dumat and Staunton, 1999). The influence of the bioturbation on the caesium distribution was also considered (Müller-Lemans and van Drop, 1995; Tyler *et al.*, 2001). Generally, sorption of caesium onto soil particles is relatively fast and strong. (Dunnigan and Francis, 1972; Sawhney, 1972).

In 1973, Ritchie and McHenry described the vertical distribution of caesium-137 in cultivated soil. They found that the depth distribution of  $^{137}\text{Cs}$  in tilled soil is different from untilled soil.

In 1974, Ritchie *et al.* used  $^{137}\text{Cs}$  measurement to study soil erosion. They compared the loss of fallout  $^{137}\text{Cs}$  with the results of soil loss obtained using the USLE (Universal Soil Loss Equation). They found that there is an exponential relationship between soil loss and  $^{137}\text{Cs}$  loss. In 1977, Ritchie and McHenry presented the results of measurements of  $^{137}\text{Cs}$  in some watersheds in the United States of America. They found that it is possible to build models to predict the distribution of caesium in small watersheds. The distribution of  $^{137}\text{Cs}$  in a small watershed was also investigated by Carlson (1978). He concluded that the loss of  $^{137}\text{Cs}$  from the watershed was small and the redistribution of  $^{137}\text{Cs}$  between the different compartments of the watershed was small.

In 1981, Brown *et al.* (1981a; 1981b) published their results of using  $^{137}\text{Cs}$  as an erosion indicator. They confirmed that the depth of the presence of  $^{137}\text{Cs}$  in a soil profile is rather a result of sedimentation than of downward displacement of  $^{137}\text{Cs}$  by leaching or bioturbation. Their results indicate that a considerable movement of soils occurs within the cultivated field. Similar results obtained McHenry and Bubenzer (1985) for White Clay Lake Watershed.

Although  $^{137}\text{Cs}$  has been used to study soil erosion, only erosion induced by water was determined. The first attempt to use  $^{137}\text{Cs}$  to study wind erosion was made at the beginning of 1990s (Sutherland *et al.*, 1991). The number of attempts to use  $^{137}\text{Cs}$  to study wind erosion in comparison to using  $^{137}\text{Cs}$  to study water erosion is small (Yan *et al.*, 2001; Li *et al.*, 2005). Caesium-137 was also used to estimate tillage redistribution of soil (Lobb *et al.*, 1995; Quine *et al.*, 1999; Schuller *et al.*, 2003).

In 1991, Walling and Quine published the results of a study of soil erosion on arable land in the United Kingdom. The importance of this publication is that it considered the limitations of this method. In particular, attention to the Chernobyl fallout must be paid. It is quite important because one of the main assumptions is that the initial distribution of  $^{137}\text{Cs}$  fallout over the study area is uniform. It is true for a global fallout but not for the Chernobyl fallout. It should be mentioned that global fallout was deposited over a long period compared to the Chernobyl fallout that was deposited during a short

period such as one rainfall. The spatial distribution of Chernobyl caesium is strongly spatially variable and thus complicates the use of caesium as a tracer of soil erosion (Higgitt *et al.*, 1992; Stach, 1996b; Lettner *et al.*, 2000; Dubois and Bossew, 2003; Renaud *et al.*, 2003). The problems connected with the Chernobyl input of  $^{137}\text{Cs}$  were discussed by De Roo (1991). The author focused on the problem of separating the Chernobyl fallout from global caesium fallout.

Caesium is used to study soil erosion also in conjunction with other radioisotopes, both natural and anthropogenic (Wallbrink and Murray, 1996). Wallbrink and Murray (1993) used the fallout radionuclides ( $^{137}\text{Cs}$ ,  $^7\text{Be}$  and  $^{210}\text{Pb}$ ) to identify erosion processes. Their analysis revealed a different penetration characteristic of those radioisotopes. Dyer and Olley (1999) used  $^{137}\text{Cs}$  to determine the origin of a recent sediment. Dyer and Olley (1999) and Wallbrink and Murray (1996) studied the grain size effect associated with the adsorption of  $^{137}\text{Cs}$  by soil minerals. Their study provided quantitative information of the grain size behaviour of fallout caesium in soil.

Du *et al.* (1998) considered the influence the  $^{137}\text{Cs}$  distribution in soil profiles on the soil erosion estimation. They showed that erosion rates may be overestimated or underestimated without considering the depth distribution pattern.

Ritchie and McCarty (2003) confirmed that carbon concentration of upland soils is significantly correlated with  $^{137}\text{Cs}$ . Their data suggest that  $^{137}\text{Cs}$  patterns may be used to help understand soil carbon dynamics on the landscape. The authors suggest that the strong relationship between  $^{137}\text{Cs}$  and soil carbon concentration is due to moving along similar physical pathways. The correlation of caesium-137 with the organic carbon was also mentioned by Takenaka *et al.* (1998).

Ritchie and McHenry in 1990 presented their review paper about the caesium technique to study soil erosion and sediment accumulation.

In 2003, Bacchi *et al.* presented their results of comparing the  $^{137}\text{Cs}$  method with USLE and WEPP models. In 2003, Porto *et al.* presented their results of validating some assumptions associated with using caesium-137 to study soil erosion. They carried out their measurements on experimental plots. There are also some papers related the limitations of the  $^{137}\text{Cs}$  measurements to study soil erosion in general (Zapata and Garcia-Agudo, 2000; Walling, 1998) as well as in particular environment (Chappell *et al.*, 1998; Chappell, 1999; Golosov, 2003) and some are concerned with mathematical modelling (Walling and Quine, 1991; Walling and He, 1999; Yang *et al.*, 1998; Zhang *et al.*, 1999).

#### 4. THE HISTORY OF USING $^{137}\text{Cs}$ AS AN EROSION TRACER IN POLAND

The first reported attempt of using of caesium-137 as a soil erosion tracer was in the middle of 1980's (Higgitt *et al.*, 1992; Froehlich and Walling, 1992). It should be mentioned that the first measurement of fallout radioisotopes in soil in Poland was a little bit earlier. Śmierzchalska

(1973) had studied the behaviour of  $^{137}\text{Cs}$  in soil, its sorption on soil particles and behaviour of caesium in soil-plants systems. The uptake of strontium-90 and caesium-137 were investigated by Glabiszewski *et al.* (1979) and Kiepuł *et al.* (1984).

Froehlich *et al.* (1993) measured  $^{137}\text{Cs}$  activity to study soil erosion and sediment delivery within the Homerka catchment in the Polish Flysh Carpathians. This work was carried out in an experimental catchment. The classical monitoring for this study area had been applied for 15 years. The authors started their measurements in this catchment in 1984. In 1986, due to the Chernobyl accident, the  $^{137}\text{Cs}$  inventories in soil in the study area significantly increased and caused problems with a comparison of samples analyzed before and after the accident. The authors concluded that caesium measurement was valuable for assessing the sediment movement for the previous 35 years. Higgitt *et al.* (1992) and Froehlich *et al.* (1993) presented the problem of using the radiocaesium to study soil erosion for an area contaminated by Chernobyl caesium. They used the  $^{134}\text{Cs}/^{137}\text{Cs}$  ratio to estimate bomb-test fallout  $^{137}\text{Cs}$  for the study area. They found that the variability of Chernobyl caesium fallout increases the uncertainty in the estimation of soil erosion and complicates the interpretation of caesium measurements. They also found that  $^{137}\text{Cs}$  as a sediment tracer offers considerable potential for assessing information about sediment mobilization and transfer within a fluvial system. Chelmicki *et al.* (1992) used  $^{137}\text{Cs}$  to study slope processes in the catchment Dworski Potok in the Carpathian Foothills. They found that the middle part of the slope was deteriorated due to water erosion much more than other parts of the slope. They also concluded that on the top of the slope there was no soil erosion or it was very small. In addition, they measured the caesium activity along the small watercourse. The purpose of this study was to establish the depth distribution of  $^{137}\text{Cs}$  in a sediment in the potential accumulation area. In 1995, Chelmicki *et al.* measured the  $^{137}\text{Cs}$  activity in soil in Carpathian foot-hills. They intended to evaluate the value and horizontal difference of  $^{137}\text{Cs}$  contamination of soil in this area and compare those results with the results for other parts of Poland. They found a strong spatial difference in activity of  $^{137}\text{Cs}$  in soil even on a small scale. They concluded that this was due to variability in precipitation during the time directly after the Chernobyl accident. The small variability in physicochemical soil properties confirms this suggestion.

Caesium-137 was used also to investigate the modern sediments in the Kłodzko Valley (Bluszcz and Śnieszko, 2000). The authors used this method to study the dilluvium and fen soil in the Nysa Kłodzka river. They found that the thickness of presence of caesium in the studied core sediments is 50-60 cm. The results confirmed that the rate of sediment deposition in the study area is quite high.

Stach and Podsiadłowski (2001) used the  $^{137}\text{Cs}$  method to measure wind soil erosion. They measured the wind erosion on a relatively large agricultural field (about 64 ha) and they obtained an average annual value of wind erosion of ca 27.7 Mg/ha yr for the study area. Unfortu-

nately, this value is probably an overestimation of wind erosion; this is because they did not consider the influence the selective sorption of caesium onto soil particles. Moreover, they used a linear relationship between the loss of caesium and soil loss. This method requires good mixing of  $^{137}\text{Cs}$  in the plough layer.

A relatively large study of soil erosion by  $^{137}\text{Cs}$  measurement was carried out by Zgłobicki (2002) for North-West part of Lubelska Highland. The study area is characterized by high value of soil erosion. This area is mainly covered by loess soil and it has a high percentage of agricultural fields in the total soil usage. Apart from  $^{137}\text{Cs}$  measurement, other measurements such as physicochemical properties of soil were also carried out. Moreover, it was found that for some soil samples the activity of  $^{134}\text{Cs}$  was found.  $^{134}\text{Cs}$  could be a valuable tracer of Chernobyl caesium but nowadays its activity in natural environment is below detection limit. To calculate the measurement data he used the simplified mass balance model provided by Zhang *et al.* (1990). Zgłobicki found that the horizontal scheme of soil erosion is very complicated. For a single slope, there are areas where erosion and accumulation of soil occur with a different intensity.

As was mentioned at the beginning of this section, there are also many articles about  $^{137}\text{Cs}$  measurements in soil or sediment but for other purposes. Many papers describe the activity of caesium in soil as a source of contamination (Szewczyk, 1994; Chibowski *et al.*, 1994). Strzelecki *et al.* (1994) described in detail the concentration of caesium in Poland. The authors compared the average concentration of caesium for all provinces in Poland. Zygmunt *et al.* (1998) and Chibowski *et al.* (1999) studied the behaviour of the  $^{137}\text{Cs}$  in soil in the vicinity of Lublin. Zygmunt *et al.* (1998) concluded that caesium does not exist in soil in ionic form; this means that only a small amount of caesium can be assimilated by plants (Zygmunt *et al.*, 1998). Chibowski *et al.* (1999) found that almost all caesium is irreversibly adsorbed in the lessive soil and the podzol soil. On the other hand, significant amount of caesium is reversibly bounded on in the brown soil. Some authors used caesium as a tool to study lake sediments (Fedorowicz *et al.*, 2002; Zalewski *et al.*, 1995).

## 5. MODELS FOR CONVERSION OF THE $^{137}\text{Cs}$ DATA INTO SOIL EROSION RATE

### *Cultivated areas*

There are many relationships to obtain quantitative estimates of soil erosion and/or deposition from  $^{137}\text{Cs}$  measurements. In a few words, usually to obtain the quantitative value of soil erosion from caesium measurements the initial input of fallout of  $^{137}\text{Cs}$  has to be known. Luckily, this input may be obtained for the area under study by relatively simple measurements of  $^{137}\text{Cs}$  in soil in places without erosion or accumulation. Generally, the main idea of using  $^{137}\text{Cs}$  measurements to study soil erosion is a comparison of the  $^{137}\text{Cs}$  inventory for studying places with the local  $^{137}\text{Cs}$  initial input. All approaches can be divided into two main groups: empirical relationships and theoretical models. Empirical equations were established to explain

the relationship between  $^{137}\text{Cs}$  loss and soil loss. This approach demands simultaneous measurement of  $^{137}\text{Cs}$  loss from erosion plots (or increase for accumulation plots) and soil loss. At present, a few empirical relationships exist (Menzel, 1960; Rogowski and Tamura, 1970a). Usually the relationship between soil loss and  $^{137}\text{Cs}$  loss is exponential.

Although these approaches are relatively easy to use, this kind of measure of soil erosion has many limitations. First, these empirical equations were established for relatively small experimental plots (Ritchie *et al.*, 1974; Ritchie and McHenry, 1975) and thus the use for other plots is rather difficult. Second, this equation is the same for undisturbed and cultivated areas. This can lead to an underestimation of soil erosion for cultivated fields and an overestimation of soil erosion for undisturbed sites. This is because the different depth distribution of  $^{137}\text{Cs}$  in soil profile for undisturbed and cultivated field (Ritchie and McHenry, 1973b; Walling and Quine, 1990; Poręba *et al.*, 2003).

The second group of approaches, to convert the  $^{137}\text{Cs}$  activity data into soil erosion value, are theoretical models. Generally, all existing theoretical models could be divided into two main groups: for undisturbed and for agricultural areas. There are two kinds of theoretical models to convert the  $^{137}\text{Cs}$  data for the agricultural fields: proportional models and mass balance models. The proportional models were established and used by many scientists (de Jong *et al.*, 1983; Kachanoski, 1987). The proportional models were summarised by Walling and Quine (1990). Brown *et al.* (1981b) established the gravimetric model but this model is a kind of proportional models. There are a few different proportional models for the calculation of the value soil loss but the differences between them are rather small. Those models are based on the assumption that  $^{137}\text{Cs}$  is completely mixed within the plough layer. The soil loss is directly proportional to the  $^{137}\text{Cs}$  loss from the soil profile. The equation can be written as follows (Walling and Quine, 1990):

$$Y = 10 \frac{BdX}{100T} \quad (5.1)$$

where:

Y – mean annual soil loss ( $\text{t ha}^{-1}\text{yr}^{-1}$ ),  
 B – bulk density of the soil ( $\text{kg m}^{-3}$ ),  
 d – the depth of the plough layer (m),  
 X – percentage reduction in total  $^{137}\text{Cs}$  inventory

(defined as  $\frac{A_{\text{ref}} - A}{A_{\text{ref}}} \cdot 100$ ),

$A_{\text{ref}}$  – the local  $^{137}\text{Cs}$  reference inventory ( $\text{Bq m}^{-2}$ ),  
 A – the measured  $^{137}\text{Cs}$  inventory at the sampling point ( $\text{Bq m}^{-2}$ ),  
 T – the time elapsed since initiation of  $^{137}\text{Cs}$  accumulation (yr).

This approach has several limitations. First, this model assumes that caesium is uniformly distributed in the plough layer. Immediately after the fallout, the surface contained more caesium than lower soil layer due to

agricultural mixing, e.g. through ploughing. It means that the results of calculation of the soil loss may be overestimated (Walling and Quine, 1990). Moreover, due to a selective removal of fine soil particles the results can be overestimated. This is because the  $^{137}\text{Cs}$  in more readily adsorbed on the fine soil fraction. In a nutshell, this approach was used many times due to its simplicity although the results are sometimes inaccurate.

To overcome the limitations of the proportional models (and also of the gravimetric approach) the mass balance model was proposed by Kachanoski and de Jong (1984). The authors supposed that the relationship between soil erosion and  $^{137}\text{Cs}$  remaining is time-dependent. They assumed that relationship between the changing of  $^{137}\text{Cs}$  in soil and soil erosion depends on atmospheric deposition of  $^{137}\text{Cs}$  and tillage dilution of  $^{137}\text{Cs}$ . The authors also considered the variation in soil erosion and  $^{137}\text{Cs}$  deposition during a year. They proposed a mass balance model as follows:

$$\frac{\partial A_t}{\partial t} = D_t - E_t K_2 C_t - K_1 A_t \quad (5.2)$$

where:

- $A_t$  – the total  $^{137}\text{Cs}$  in plough layer ( $\text{Bq m}^{-2}$ );
- $t$  – time;
- $D_t$  – the average atmospheric deposition rate  $^{137}\text{Cs}$  ( $\text{Bq m}^{-2} \text{ yr}^{-1}$ );
- $E_t$  – the average soil erosion rate ( $\text{kg m}^{-2} \text{ yr}^{-1}$ );
- $K_1$  – the radioactive decay constant for  $^{137}\text{Cs}$  ( $0.023 \text{ yr}^{-1}$ );
- $K_2$  – enrichment coefficient
- $C_t$  – concentration of  $^{137}\text{Cs}$  in the plough layer ( $\text{Bq kg}^{-1}$ ).

The enrichment coefficient  $K_2$  was assumed because the  $^{137}\text{Cs}$  concentration in eroded sediment were about 10 % higher than  $^{137}\text{Cs}$  concentrations in the source soil (Kachanoski and de Jong, 1984).

This model takes into account the current atmospheric deposition of  $^{137}\text{Cs}$  and the dilution of the  $^{137}\text{Cs}$  by tillage as well. The authors concluded that this model is probably limited to sites where soil erosion is between 0.5 and 10  $\text{kg m}^{-2} \text{ yr}^{-1}$ . They also suggested that outside of this limits the relative errors are higher.

In 1990, a simplified version of the mass balance model was published by Zhang *et al.* (1990). They assumed that the total  $^{137}\text{Cs}$  fallout occurred in 1963, rather than in the period from 1954 to 1976. The mean annual soil loss rate can be expressed as follows:

$$Y = 10dB \left[ 1 - \left( 1 - \frac{X}{100} \right)^{1/(t-1963)} \right] \quad (5.3)$$

where:

- $Y$  – mean annual soil loss ( $\text{t ha}^{-1} \text{ yr}^{-1}$ );
- $d$  – depth of the plough layer (m);
- $B$  – bulk density of soil ( $\text{kg m}^{-3}$ );
- $X$  – the percentage reduction in total  $^{137}\text{Cs}$  inventory (defined as before – see Eq. 5.1
- $t$  – time since the 1963.

Although this model is very easy to use, the main assumption of this approach that the total  $^{137}\text{Cs}$  fallout input occurred in 1963 seems to be an oversimplification. This model does not take into account the value of  $^{137}\text{Cs}$  freshly removed from soil surface before incorporation into the plough layer by ploughing. The problem with selective sorption of  $^{137}\text{Cs}$  by soil particles could be solved by adding a particle size correction factor (Zhang *et al.*, 1999), but still the main assumption of this model, especially for areas contaminated by Chernobyl caesium is improper.

To overcome the problem with selective sorption and also removing of the fresh deposition of  $^{137}\text{Cs}$  before mixing by ploughing the improved mass balance model was established as follows (Walling and He, 1999):

$$\frac{dA(t)}{dt} (1 - \Gamma) I(t) - \left( \lambda + P \frac{R}{d} \right) A(t) \quad (5.4)$$

where:

- $A(t)$  – cumulative  $^{137}\text{Cs}$  activity per unit area ( $\text{Bq m}^{-2}$ );
- $R$  – soil erosion rate ( $\text{kg m}^{-2} \text{ yr}^{-1}$ );
- $d$  – the average plough depth ( $\text{kg m}^{-2}$ );
- $\lambda$  – the radioactive decay constant for  $^{137}\text{Cs}$  ( $\text{yr}^{-1}$ );
- $I(t)$  – annual  $^{137}\text{Cs}$  deposition flux ( $\text{Bq m}^{-2} \text{ yr}^{-1}$ );
- $\Gamma$  – the percent of the fresh deposition of  $^{137}\text{Cs}$  removed by erosion before mixing into the plough layer;
- $P$  – the particle size correction factor.

According to He and Walling (1997), if an exponential distribution of the  $^{137}\text{Cs}$  in a soil profile for the initial distribution of  $^{137}\text{Cs}$  fallout could be assumed,  $\Gamma$  can be expressed as:

$$\Gamma = P\gamma \left( 1 - e^{-R/H} \right) \quad (5.5)$$

where:

- $\gamma$  – the proportion of the annual  $^{137}\text{Cs}$  input susceptible to removal by erosion and depends on the local agricultural and rainfall conditions;
- $H$  – the relaxation mass depth of the initial distribution of fallout  $^{137}\text{Cs}$  in the soil profile ( $\text{kg m}^{-2}$ ). The factor  $H$  could be established experimentally for particular location.

To establish the  $P$  value for the study area information of grain size distribution for soil sediment and the original soil is required. He and Walling (1996) established the relationship between  $P$  value and specific surface area of the sediment and of the soil. The problems connected with the particle size effect and particle size correction factor are also discussed by Porto *et al.* (2003).

This mass balance model is more realistic compared to the previous mass balance model and the simplified mass balance model. This model requires apart from the plough depth two additional parameters, namely, the relaxation mass depth and parameter  $\gamma$ . This is also an additional source of uncertainty in soil erosion estimation.

In 2000, Yang *et al.* presented a quantitative model for estimating soil loss. This model is of the mass balance type. The authors considered the remaining fraction of the surface enrichment layer, the thickness of the surface enrich-

ment layer and also the plough layer, annual  $^{137}\text{Cs}$  fallout. The main problem is that this model requires an information about the behaviour of  $^{137}\text{Cs}$  in the top soil layer as well as the model described earlier. Particularly this model requires information about the remaining fraction of the surface remaining layer and the thickness of the surface enrichment layer. It is usually difficult to establish exact value of those parameters.

For cultivated areas, not only water erosion of soil occurs but also a tillage redistribution (Govers *et al.*, 1994). Probably the first attempt to use  $^{137}\text{Cs}$  as a tracer of tillage erosion was reported by Lobb *et al.* (1995). A useful model where the tillage redistribution is incorporated was described by Walling and He (1999). For the point where soil erosion occurs, the variation of the total  $^{137}\text{Cs}$  inventory is expressed as follows:

$$\frac{dA(t)}{dt} = (1 - \Gamma)I(t) + R_{t,in}C_{t,in}(t) - R_{t,out}C_{t,out}(t) - R_wC_{w,out}(t) - \lambda A(t) \quad (5.6)$$

where:

$A(t)$  – cumulative  $^{137}\text{Cs}$  activity per unit area ( $\text{Bq m}^{-2}$ );  
 $C_{t,in}$  – the  $^{137}\text{Cs}$  concentration of the sediment associated with tillage input ( $\text{Bq kg}^{-1}$ );  
 $C_{t,out}$  – the  $^{137}\text{Cs}$  concentration of the sediment associated with tillage output ( $\text{Bq kg}^{-1}$ );  
 $C_{w,out}$  – the  $^{137}\text{Cs}$  concentration of the sediment associated with water output ( $\text{Bq kg}^{-1}$ );  
 $R_w$  – the water erosion rate ( $\text{kg m}^{-2} \text{yr}^{-1}$ );  
 $\Gamma$  – the percentage of the fresh deposit of  $^{137}\text{Cs}$  fallout removed by erosion before mixing into the plough layer.

The value of net soil erosion rate could be expressed as follows:

$$R = R_{t,out} - R_{t,in} + R_w \quad (5.7)$$

Similar equation could be written for the point where deposition takes place. The contribution of the tillage effect could be expressed by a downslope sediment flux as it was described by Govers *et al.* (1996). Finally, the contribution of the tillage effect depends on the slope angle below and above the sampled point and on the slope length as it was suggested by Quine (1999). The model described above is probably the most realistic from all the models but it is difficult to use. It is important to say that this model requires additional information. Moreover, this model like all models described above is sensitive to changing the additional parameters. This is a potential source of uncertainty.

#### Undisturbed areas

The rate of soil erosion for undisturbed areas is usually much smaller than for cultivated areas. Moreover due to the lack of mechanical mixing of soil during agricultural processes, the vertical distribution of  $^{137}\text{Cs}$  in soil profile is different than for cultivated areas. (He and Walling, 1997; Poręba *et al.*, 2003) Furthermore, the soil erosion processes could be different for undisturbed soil and for cultivated soils.

For undisturbed soil, the models to convert  $^{137}\text{Cs}$  measurement data into soil erosion value can be divided into two main groups: the profile distribution models and the diffusion models. Both groups of models are classified as theoretical models. There exists also an empirical relationship similar to relationships for cultivated soils, but those relationships have many limitations and thus soil erosion estimation might be incorrect (Walling and He, 1999). The profile distribution models are based on the assumption that generally for undisturbed soil the caesium activity decreases with depth exponentially (Walling and He, 1999; Walling and Quine, 1990):

$$A'(x) = A_{ref} (1 - e^{-x/h_0}) \quad (5.8)$$

where:

$A'(x)$  – amount of  $^{137}\text{Cs}$  above depth  $x$  ( $\text{Bq m}^{-2}$ );  
 $x$  – depth ( $\text{kg m}^{-2}$ );  
 $h_0$  – the factor describing the profile shape ( $\text{kg m}^{-2}$ ).

For the soil erosion place, according to Walling and Quine (1990) and Walling and He (1999), the relationship could be expressed by the following equation:

$$Y \frac{10}{t - 1963} \ln \left( 1 - \frac{X}{100} \right) h_0 \quad (5.9)$$

where:

$Y$  – rate of soil erosion, ( $\text{t ha}^{-1} \text{yr}^{-1}$ );  
 $t$  – year of sample collection;  
 $X$  – reduction of the  $^{137}\text{Cs}$  inventory at the sampling point relative to local reference inventory of  $^{137}\text{Cs}$ , (in percent).

For this model, it is also possible to assume such as for simplified mass balance model described in the previous chapter that the total  $^{137}\text{Cs}$  fallout occurred in 1963. This simplified version of the model was described by Walling and He (1999). It should be mentioned, that such a simplified mass balance model could be a source of the overestimation of soil erosion rate.

The models described above, although quite easy to use, might not provide correct values of soil erosion estimation. The results of soil erosion calculation depend on the distribution pattern of  $^{137}\text{Cs}$  in the soil profile, as was shown by Yang *et al.* (1998). They performed a numerical simulation of the soil erosion rate by using different values of  $^{137}\text{Cs}$  loss as well as depth distribution of  $^{137}\text{Cs}$ . They found that for a small percentage, the reduction in the total  $^{137}\text{Cs}$  inventory, the relationship between soil loss and  $^{137}\text{Cs}$  loss is almost linear. They concluded that the depth distribution patterns of  $^{137}\text{Cs}$  are a major factor for estimating the rate of soil loss. To carry out more realistic studies, the diffusion models are better (He and Walling, 1997; Walling and He, 1999). The behaviour of caesium in soil after the deposition is quite complex and the depth distribution of  $^{137}\text{Cs}$  in soil profiles is time-dependent (He and Walling, 1997; Walling and He, 1999; Owens and Walling, 1998). There are physical, chemical and biological processes responsible for caesium redistribution within soil. The vertical movement of caesium within a soil can be described by the following equation:

$$\frac{\partial C_u(x,t)}{\partial t} = D \frac{\partial^2 C_u(x,t)}{\partial x^2} - V \frac{\partial C_u(x,t)}{\partial x} - \lambda C_u(x,t) \quad (5.10)$$

where:

$C_u(x,t)$  – the  $^{137}\text{Cs}$  concentration ( $\text{Bq kg}^{-1}$ ) in soil

$x$  – cumulative mass depth m,

$t$  – time

$D$  – the effective diffusion coefficient ( $\text{kg}^2 \text{m}^{-4} \text{yr}^{-1}$ ),

$V$  – the migration rate ( $\text{kg m}^{-2} \text{yr}^{-1}$ ).

The depth of penetration of  $^{137}\text{Cs}$  depends on  $D$  and  $V$  values.

The solution of equation (5.9) for non-eroding sites as well as for areas where soil erosion and deposition occurs were presented by Owens and Walling (1998) and Walling and He (1999). Owens and Walling (1998) also considered the influence of particle size composition. That is because these values could be different for mobilised soil and for source soil. The limitations of this model are similar to the limitations of the mass balance models for cultivated land. This model also requires additional information usually established empirically.

## 6. CONCLUSIONS

Currently, Caesium-137 is widely used to study soil redistribution. This method is very useful and it has many advantages. Firstly, this method allows obtaining the average value of soil erosion for medium-term (about 40 years). Secondly, this method also allows obtaining the net value of soil erosion. This is particularly important because it is well known that a part of soil sediment remains in catchment. Moreover, compared to the traditional methods, this method requires only a single sampling. Traditional methods require significantly longer measurements.

The caesium method has also some limitations. Firstly, to obtain quantitative value of soil erosion it is important to know the reference value of  $^{137}\text{Cs}$  inventory for the study area and sometimes it is difficult to find a reference place for the study area. It should be noted that for areas contaminated by “Chernobyl caesium” the application of this method may be difficult. Secondly, the calculation of value of soil erosion and deposition strongly depends on the model used; the models are sensitive to the parameter choice. In other words, it is essential to use models and determine the parameters for those models with great care. Probably to overcome the limitations of the method, the solution is to take into consideration other isotopes like  $^{210}\text{Pb}$  or  $^7\text{Be}$  together with  $^{137}\text{Cs}$ . There are publications suggesting that this approach has a future (Olley *et al.*, 1993; Wallbrink and Murray, 1996). Finally, the  $^{137}\text{Cs}$  method to study soil erosion is relatively fast and cheap compared to the traditional.

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