SESSION 2

PRESERVING AND PROTECTING SOIL RESOURCES

Up-scaling the Application of Fallout Radionuclides to Support Catchment Sediment Management Programmes

D.E. Walling[1,*], P. Du[2], P. Porto[3] and Y. Zhang[4]

ABSTRACT

The application of fallout radionuclides in soil erosion and sedimentation investigations has provided a valuable tool for improving the understanding of erosion and sediment transfer processes. However, most studies using fallout radionuclides have focussed on small areas. Increasing acceptance of the important role of fine sediment in degrading aquatic habitats and in the transfer and fate of nutrients and contaminants within terrestrial and fluvial systems has emphasized the need for sediment management programmes in many areas of the world. Such programmes commonly focus on the catchment scale and an understanding of the sediment budget of a catchment represents a key requirement for developing effective sediment management strategies. Although the need for information on catchment sediment budgets is clear, obtaining such information faces many practical problems. While fallout radionuclides offer considerable potential for use in documenting sediment budgets for small areas, there is a need to up-scale their use to provide information at the catchment — and larger scales. To date there have been few attempts to develop and implement such upscaling. This paper describes the results of three studies undertaken by the authors to address this need. These examples include, firstly, the use of caesium-137 (^{137}Cs) measurements to provide the data required for a national scale soil erosion inventory; secondly, the use of areawide-scale spatially distributed sampling of ^{137}Cs inventories to assemble information on catchment sediment budgets; and finally the use of ^{137}Cs and excess lead-210 (^{210}Pb$_{ex}$) measurements to establish sedimentation rates on river floodplains.

Key words: *fallout radionuclides, caesium-137, soil erosion, floodplain sedimentation, catchment-scale, sediment budget.*

INTRODUCTION

In recent years, the potential for using fallout radionuclides to provide an improved understanding of erosion and sediment transfer processes has been clearly and successfully demonstrated (e.g. Zapata, 2002). Particular advantages of the approach include the ability to obtain retrospective spatially distributed data on the basis of a single site visit and without the need to install permanent monitoring equipment and structures (e.g. Walling and Quine, 1995; Mabit, Benmansour and Walling, 2008). The conjunctive use of different fallout radionuclides can provide a basis for assembling information relating to different time windows (Mabit, Benmansour and Walling, 2008), and the ability to generate spatially distributed data has coincided with the increasing need for such data for validating physically-based distributed soil erosion models (He and Walling, 2003). There has been some criticism of the approach directed at the lack of validation of results obtained (Parsons and Foster, 2011), but this ignores the very substantial and convincing body of empirical validation that has now been reported (e.g. Porto, *et al.*, 2003a and b; Belyaev *et al.*, 2008). To date, however, most work in applying fallout radionuclides in soil erosion and sedimentation investigations has focussed on relatively small areas, such as individual fields or sedimentation sites, although there have been some attempts to look at larger areas (e.g. Mabit, Bernard and Laverdière, 2007). The focus on small areas has meant that there have been few attempts to use fallout radionuclides in larger scale sediment budget investigations linked to catchment management and the design and implementation of sediment control strategies for individual catchments. The key role of fine sediment in degrading aquatic ecosytems and in the transfer and fate of nutrients and contaminants in terrestrial and fluvial systems is being increasingly recognized and this means that sediment management is assuming growing importance in many areas of the world, including both areas with high erosion rates and sediment yields and those where such rates and yields are very much lower. The latter areas are frequently the most sensitive to small changes in soil and sediment redistribution rates and represent areas where visual evidence of sediment mobilization and transfer is often limited. It is frequently these areas where fallout radionuclides are most useful. As demonstrated by Walling and Collins (2008) and Gellis and Walling (2011), the catchment sediment budget is commonly the key to designing an effective sediment strategy, since it provides information on the relative importance of different sediment sources, on the deposition and storage of sediment as it moves through the catchment sediment system and on the relationship between the sediment output and internal sources and sinks. Fallout radionuclides are often employed to provide key information needed to construct a catchment budget, and without such information the task of establishing a sediment budget can prove very difficult, if not impossible.

As indicated above, the use of fallout radionuclides to provide information on the functioning of a catchment sediment budget

1 Geography, College of Life and Environmental Sciences, University of Exeter, Exeter, Devon, EX4 4RJ, UK

2 International Research and Training Center in Erosion and Sedimentation (IRTCES), P. O. Box 366, Beijing 100048, P. R. China

3 Dipartimento STAFA, Universersità degli Studi Mediterranea, Località Feo de Vito, Reggio Calabria, Italy

4 ADAS, Pendeford House, Wolverhampton, WV9 5AP, UK

* E-mail address of corresponding author: d.e.walling@exeter.uk

necessitates an up-scaling of the approach. It is essentially impossible to extend the detailed sampling commonly applied to an individual field across an entire catchment, due to cost and other practical constraints. Even if a large team is available to collect samples, the cost and time involved in analysing large numbers of samples by gamma spectrometry are likely to prove insurmountable problems. As a result, there is a need to develop new approaches that permit such up-scaling without introducing insuperable obstacles. Up-scaling approaches could involve methods of extrapolating the data obtained from a small number of measurements over a much larger area or optimizing the information content of a small number of measurements. The work of Mabit, Bernard and Laverdière (2007) cited above, subdivided the 217 km^2 catchment of the River Boyer in Quebec, Canada into a series of essentially homogeneous units termed isosectors and collected representative data from those units and extrapolated these data across the individual units using a GIS framework.

The authors have also directed attention to the potential for up-scaling the use of fallout radionuclides, and particularly ^{137}Cs measurements, to larger areas, in order to provide the information required for establishing catchment sediment budgets. This paper describes the results of three studies undertaken in recent years to explore different approaches and objectives. These examples include, firstly, the use of ^{137}Cs measurements to provide data required for a national scale soil erosion inventory; secondly, the use of wide-scale spatially distributed sampling of ^{137}Cs inventories to assemble information on the sediment budgets of several small- to medium-scale catchments in Southern Italy and, finally, the use of caesium-137 (^{137}Cs) and excess lead-210 (^{210}Pb$_{ex}$) measurements to establish sedimentation rates on river floodplains, which can represent important sinks in the sediment budgets of larger catchments and therefore need to be considered when the overall perspective is up-scaled.

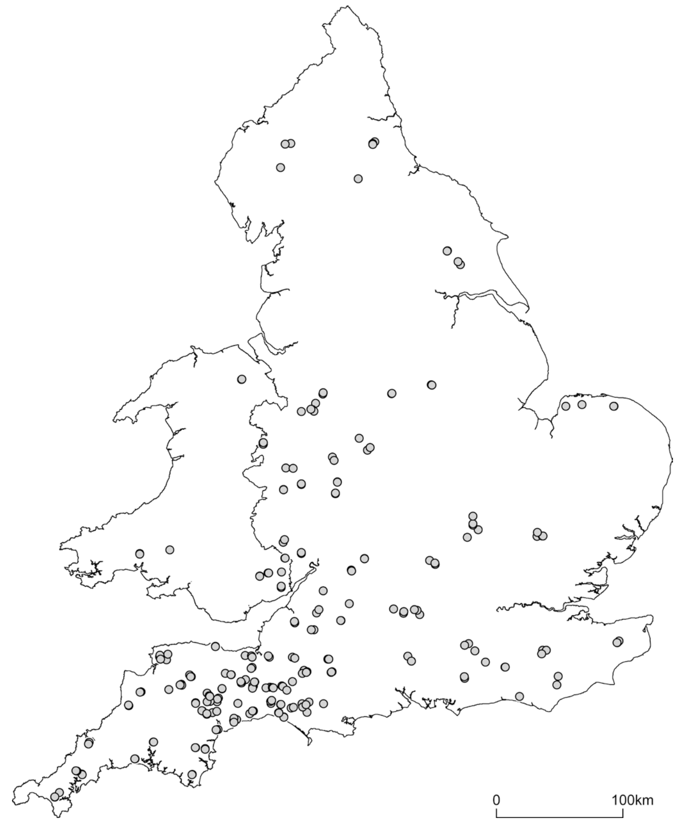

FIGURE 1. The locations selected for field sampling within the project.

A NATIONAL SCALE SOIL EROSION INVENTORY

The national soil erosion inventory study was funded by the Department for Environment, Food and Rural Affairs (DEFRA). It focussed on agricultural land in England and Wales and aimed to use ^{137}Cs measurements to provide national-scale information on the magnitude of soil erosion rates and the sustainability of the soil resource, as well as the importance of soil erosion from agricultural land as a sediment source. In the absence of a national network of erosion plots or other ongoing monitoring, ^{137}Cs measurement were seen as a valuable means of assembling information on mean soil erosion rates over the past ~50 yr and the influence of topography, soil type, land use and other controlling factors on soil erosion rates. The approach taken is described in more detail by Walling and Zhang (2010). In brief, there were three key elements. Firstly, preliminary fieldwork was used to confirm that the collection of a relatively small number of bulk cores (e.g. 10–12) from a small field with a simple topography using a representative transect, could provide a reliable estimate of the longer-term (50 yr) gross and net soil losses from that field using standard conversion models (Walling and He, 1999). Secondly, by sampling a substantial number of such fields, representative of a wide range of slopes and forms, soil texture, land use and hydrometeorological conditions (e.g. mean annual rainfall and hydrologically effective rainfall), and obtaining reliable estimates of

TABLE 1. Summary statistics for the erosion rates for the sample of 248 fields (t·ha^{-1}·yr^{-1})

	Arable			Pasture		
	Eroding area rate	Gross erosion rate	Net erosion rate	Eroding area rate	Gross erosion rate	Net erosion rate
Minimum	1.2	1.2	0.0	0.5	0.7	0.0
Maximum	30.3	29.3	27.3	16.0	13.0	11.7
Average	10.6	8.4	6.7	3.2	2.5	1.8
Median	8.8	6.6	5.2	2.6	1.8	1.2
1st quartile	6.5	4.7	2.0	1.7	0.9	0.2
3rd quartile	13.3	11.2	9.1	3.8	3.0	2.6
CV * (%)	80	99	140	82	118	200

* CV was calculated as: (Q1–Q3) / median × 100

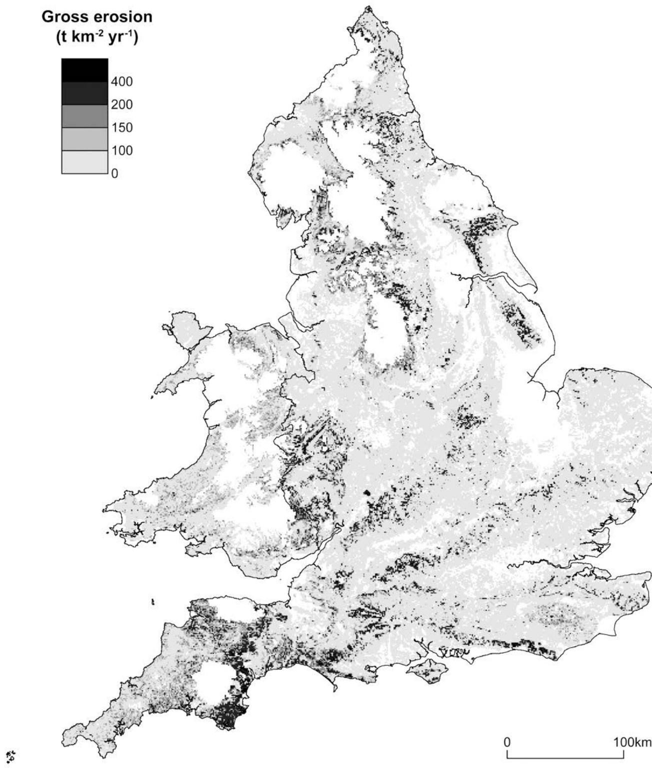

Figure 2. A national map of gross erosion rates for a combination of cultivated and pasture land use, based on spatial extrapolation of the field-scale erosion rates documented by ^{137}Cs measurements.

the reference inventory for such fields, it was possible to assemble information on the range of soil erosion rates found on agricultural land in England and Wales and their key controls. Thirdly, these data were used to populate a typology model which was employed to extrapolate the available estimates of erosion rates over the entire agricultural land area in England and Wales.

Attention was restricted to areas of agricultural land below an altitude of 300 m. National Parks, urban areas and areas of minimal slope (<1°), where water erosion was likely to be negligible, were excluded from the study. In total, 248 fields were sampled, of which 133 were arable fields and 115 were under permanent pasture (Figure 1). The ^{137}Cs measurements undertaken on the cores collected from the 248 fields were used to derive estimates of several measures of the longer-term (ca. 50 yr) soil erosion rate. These included the proportion of the field that was eroding, the erosion rate within the eroding area and the gross and net erosion rates for the field. A summary of the information on soil erosion rates provided by the 248 fields is presented in Table 1.

These data indicate that, as might be expected, both gross and net erosion rates associated with areas of arable cultivation are considerably greater than those for pasture areas. The ratio of net to gross erosion, which provides a measure of the efficiency of sediment transfer out of the field also differed between these two categories of land use and was typically > 0.6 for arable fields and < 0.6 for pasture fields, emphasizing the lower efficiency of sediment delivery from pasture areas. Comparison of the magnitude of the erosion rate estimates obtained from the study with existing erosion hazard assessments based primarily on soil texture, confirmed the broad consistency of the two measures of erosion intensity.

The typology model used to extrapolate the assembled data over the entire study area was based on detailed analysis of the key variables controlling the spatial variability of soil erosion rates at the national scale. The final model involved three primary variables, i.e. slope steepness, soil texture (percent clay, silt and sand) and land use and provided estimates of both gross and net erosion for indivdual grid cells. This model was coupled with a GIS, comprising 1 km × 1 km grid cells and incorporating a 50 m DEM (Digital Elevation Model), for use in deriving information on slope steepness, as well as information on soil texture and land use for each grid cell. The extrapolation procedure permitted estimation of the likely proportion of cultivated and pasture land in each grid cell and the erosion rate reflecting these proportions. National maps of gross and net erosion rates associated with arable, pasture and mixed land use were produced. Figure 2 presents the national map of gross erosion rates associated with the likely combination of arable and pasture land use in each grid cell. Spatial integration of the data presented on these maps was used to generate information on frequency distributions of gross and net erosion rates from agricultural land in England and Wales. These frequency distributions indicated that a soil loss tolerance of 2 $t{\cdot}ha^{-1}{\cdot}yr^{-1}$ is exceeded on ~7 percent of the agricultural land in England and Wales for gross erosion and on ~5 percent of the agricultural land for net erosion.

The maps of gross and net soil losses represent soil loss from individual fields and cannot be used directly to represent sediment inputs to river channels, which may be of primary interest when constructing a sediment budget. For this purpose, the national map of net soil loss was coupled with an equivalent map of slope-channel connectivity previously developed by the authors (Walling and Zhang, 2004), and also based on 1 km grid cells, to provide a national map of the sediment input to the local channel system. This map indicated that contributions of surface erosion from agricultural land typically ranged between 5 and 50 $t{\cdot}km^{-2}{\cdot}yr^{-1}$ and these values are consistent with available data on suspended sediment yields for catchments in England and Wales. These yields will include contributions from sources other than the surface of agricultural land such as channel or bank erosion, and can thus be expected to be significantly greater than those associated solely with mobilization of sediment from agricultural land.

USE OF SPATIALLY DISTRIBUTED SAMPLING TO DERIVE INFORMATION ON CATCHMENT SEDIMENT BUDGETS

The spatially intensive ^{137}Cs sampling programmes required to assemble information on soil redistribution rates for use in establishing the sediment budget of a medium-sized, or even small, catchment using standard approaches will commonly prove impractical in terms of the vast number of samples that need to be collected and assayed. An alternative wide-scale spatially distributed sampling strategy has been proposed by the authors (see Porto, Walling and Callegari, 2011). Collection of cores for ^{137}Cs measurement from a substantial number of representative points distributed throughout a catchment and using the associated values of ^{137}Cs inventory to estimate soil redistribution rates at the sampling points can provide a representative sample of soil and sediment redistribution rates within the catchment. When combined with information on the sediment output from the catchment, this information can provide a basis for establishing a sediment budget. This approach was applied successfully by the authors in three small- to medium-scale catchments in southern Italy (Porto, Walling and Callegari, 2011).

The three catchments are illustrated in Figure 3 and further information on their characteristics is provided in Table 2. Their drainage areas ranged from 0.015 km^2 to 31.61 km^2 and this substantial range was seen as providing a means of assessing the impact of scale (i.e.

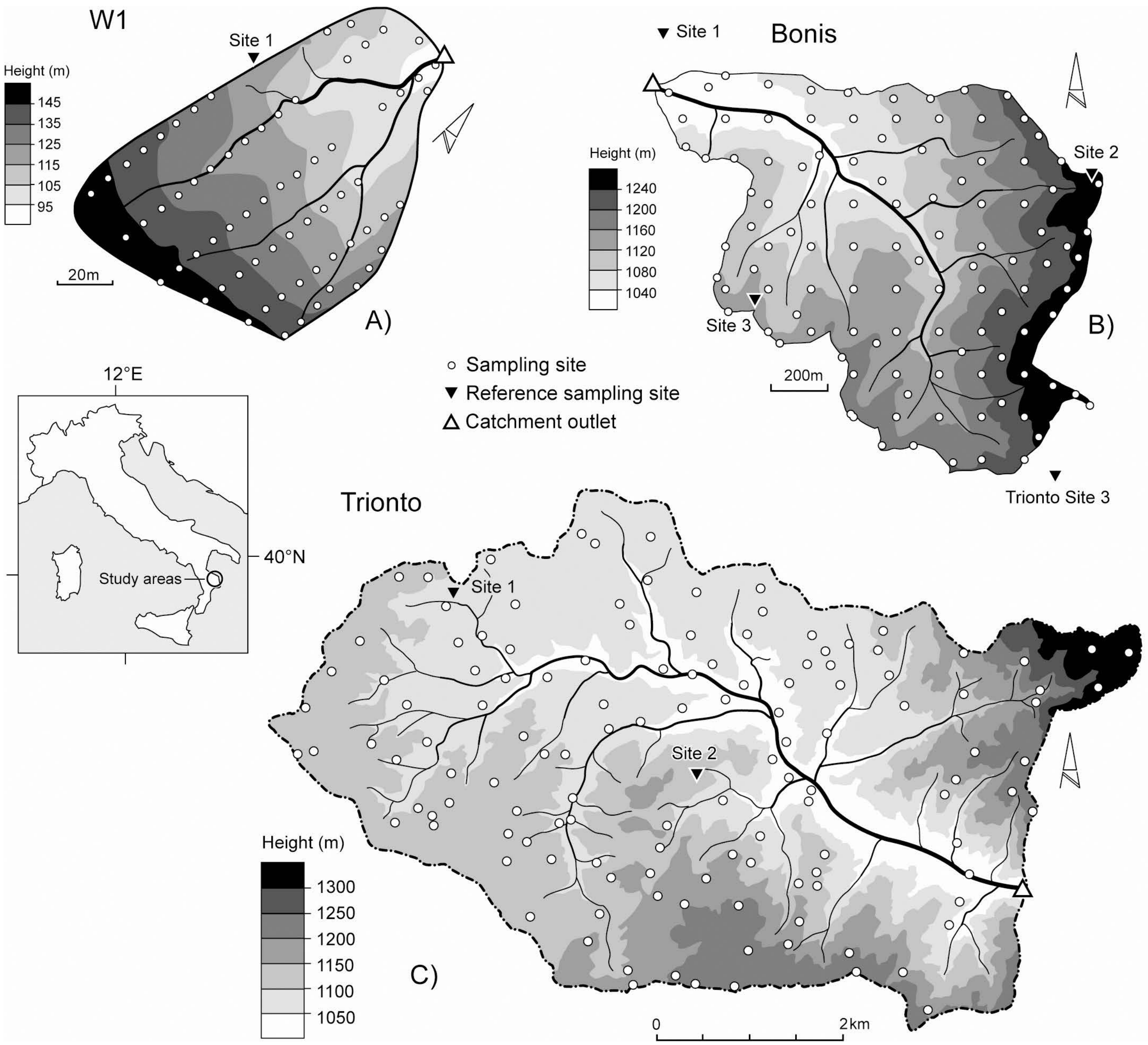

Figure 3. The study catchments: W1 (A), Bonis (B) and Trionto (C).

increasing catchment area) on their sediment budgets. Catchment W1 is uncultivated and has a rangeland cover, whereas the Bonis catchment is largely under forest and the Trionto catchment is largely cultivated. Information on long-term suspended sediment yields was available for the three catchments (see Table 2) based on long-term sediment sampling programmes (W1 and Trionto) and periodic emptying of sediment traps (Bonis).

Bulk cores were collected from each catchment for establishing the ^{137}Cs inventory at individual sampling points. Different sampling strategies were used for each catchment. In the case of catchment W1, this involved the collection of 68 bulk cores along five essentially parallel linear transects (Figure 3A). For the Bonis catchment, 55 bulked replicate cores were collected from the intersections of a 150 m × 150 m grid and these cores were supplemented by a further 55 bulked replicate cores collected from sites selected to improve the coverage of topographic variability (Figure 3B). In the case of the Trionto catchment, the larger size of the catchment and constraints on access precluded the use of a systematic sampling network, and sampling involved the selection of 128 essentially random sampling points across the catchment aimed at providing a representative coverage of the terrain and a relatively uniform spatial coverage (Figure 3C). Additional sampling was carried out in the three catchments to establish the reference inventories and to provide information on the depth distributions of ^{137}Cs in the individual catchments as required by the conversion models used to estimate soil redistribution rates from the measurements of ^{137}Cs inventory. The considerable range of altitude associated with the Trionto catchment meant that three reference inventories, related to three altitudinal zones were established.

The values of ^{137}Cs inventory obtained for the sampling points in the catchments were used to derive estimates of soil redistribution rates using appropriate conversion models. In the case of catchment W1 and the Bonis catchment where the soils were uncultivated, a diffusion and migration model was employed. A mass balance

Table 2. Characteristics of the study catchments

Catchment	Area (km^2)	Mean altitude (m a.s.l.)	Mean slope (%)	Relief ratio	Primary land use	Sediment yield ($t \cdot km^{-2} \cdot yr^{-1}$)
W1	0.015	122	53	0.17	Rangeland	1 16
Bonis	1.39	1131	40	0.14	Forest	25.0
Trionto	31.61	1100	8.4	0.05	Cultivated	21.0

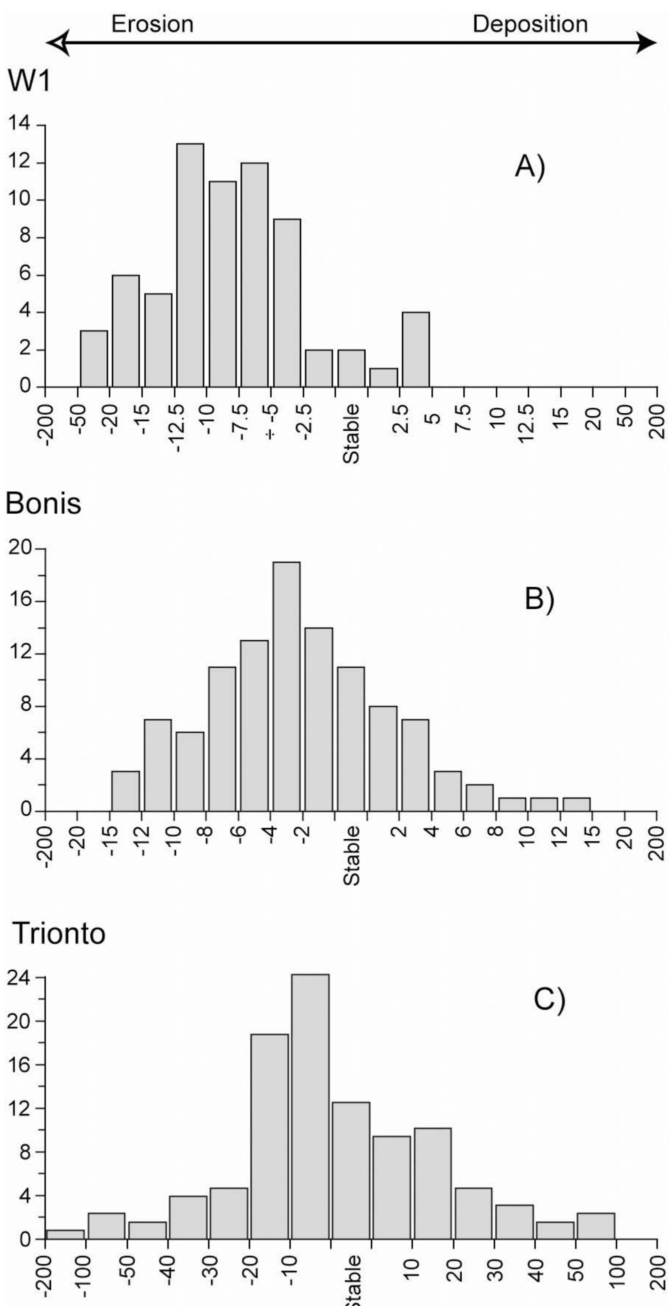

FIGURE 4. Frequency distributions of erosion and deposition rates ($t \cdot ha^{-1} \cdot yr^{-1}$) associated with the sampling points within each study catchment: (A) W1, (B) Bonis and (C) Trionto (based on Porto, Walling and Callegari, 2011).

model was employed for the cultivated areas in the Trionto catchment (Porto, Walling and Callegari, 2011). The resulting estimates of soil redistribution rate are presented in Figure 4, which depicts frequency distributions of erosion and deposition rates for the three catchments. In the case of catchment W1, these data provide little evidence of deposition and suggest that most of the mobilized sediment is transported directly to the catchment outlet. This conclusion was confirmed by field inspection, which indicated that areas of deposition were very limited. In the larger Bonis and Trionto catchments, deposition becomes progressively more important. This is a direct reflection of increasing catchment size and the associated greater opportunities for deposition. Clear contrasts in the magnitude of the soil redistribution rates are also apparent between the three catchments. Redistribution rates, and particularly erosion rates, are, as might be expected, generally lowest in the forested Bonis catchment and highest in the cultivated Trionto catchment, with catchment W1, supporting a rangeland cover, falling between the two.

If the frequency distributions of soil redistribution rates presented in Figure 4 are assumed to provide representative information on medium-term average rates of erosion and deposition on the slopes of the individual study catchments, they can be used to estimate the gross erosion from the slopes and the on-slope deposition and thus the ‘slope’ component of the sediment budget. The availability of estimates of mean annual sediment yield provides another key component of the sediment budget, namely the output. If the slope component is assumed to represent the input to the channel system from the slopes, the difference between this and the sediment yield at the catchment outlet can be attributed to deposition during conveyance through the channel system. This will provide a minimum estimate of such conveyance losses, since the amount of sediment transported through the channel system could be greater than the input from the slopes, due to additional sediment supply from eroding channel banks. However, this source was judged to be of limited importance in the study catchments. The sediment budgets for the three study catchments established using the above information and assumptions are presented in Figure 5.

These budgets emphasize the increasing importance of sediment deposition, both on the slopes and within the channel system, as catchment scale increases. The catchment sediment delivery ratio (SDR) (Walling, 1983) provides a useful means of assessing the efficiency of sediment delivery from a catchment, and Figure 5 indicates that the SDR declines from 98 percent for catchment W1 to 7 percent for the Bonis catchment and to two percent for the Trionto catchment. This is consistent with existing representations of the inverse relationship between SDR and catchment area (e.g. Roehl, 1962).

DOCUMENTING SEDIMENTATION RATES ON RIVER FLOODPLAINS

A different dimension of the requirement to up-scale the use of fallout radionuclides in order to provide information at the catchment scale, is the need to furnish information on key components of the sediment budgets of larger catchments. Attention must be

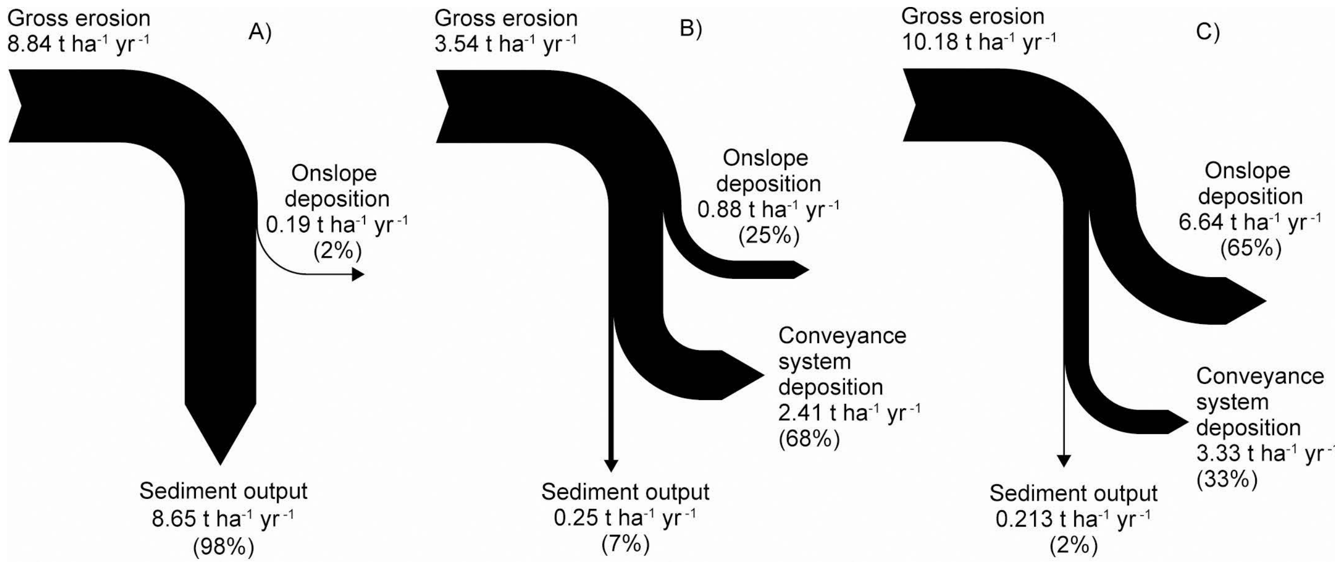

FIGURE 5. Schematic sediment budgets for catchment W1 (A), the Bonis catchment (B) and the Trionto catchment (C). The SDR associated with the sediment output from each catchment is indicated as a percentage (based on Porto, Walling and Callegari, 2011).

directed to the downstream channel system as well as the catchment slopes, since as catchment size increases, conveyance losses associated with the channel system can exert an increasing influence on the sediment output from a catchment and thus the catchment sediment budget. River floodplains are of considerable significance in this context since they can represent very important sediment sinks. The need to obtain estimates of rates of medium-term sediment accumulation or storage within floodplain systems has directed attention to the use of fallout radionuclides and more particularly ^{137}Cs and excess lead-210 (^{210}Pb$_{ex}$) measurements for this purpose. In this application, the ability to identify peaks of ^{137}Cs fallout associated with the peak of bomb fallout in 1963 and the Chernobyl fallout in 1986 (where such fallout occurred) can provide valuable time markers. Because of its essentially continuous fallout, ^{210}Pb$_{ex}$ does not provide peaks of a known age, but information on the downcore decline in ^{210}Pb$_{ex}$ activity can be used to derive estimates of recent sedimentation rates, based on the known half-life of ^{210}Pb of 22.3 yr (Du and Walling, 2012). By studying the depth distribution of both ^{137}Cs and ^{210}Pb$_{ex}$ in floodplain cores, it is possible to obtain estimates of the average sedimentation rate over the past ca. 100 yr and for specific sub-periods. If such data can be spatially extrapolated across a river floodplain, by collecting representative cores in different locations it is possible to derive estimates of the mass of sediment sequestered in a floodplain system and thus establish its importance as a sediment sink within the catchment sediment budget and assess any changes in that importance through recent time.

Figure 6 provides an example of this potential. The data presented relate to a core collected at Welshpool, UK in 2009. In this case, the ^{137}Cs profile provides clear evidence of the 1963 peak of bomb fallout at a mass depth of ~50 g/cm^2 and the 1986 Chernobyl peak at a depth of ~17 g/cm^2. The ^{210}Pb$_{ex}$ profile was used to derive an estimate of the mean annual sedimentation rate over the past ~60 yr and thus back to 1950. These estimates indicate an average sedimentation rate of 1.14 g·cm^{-2}·yr^{-1} over the period 1950–2009, based

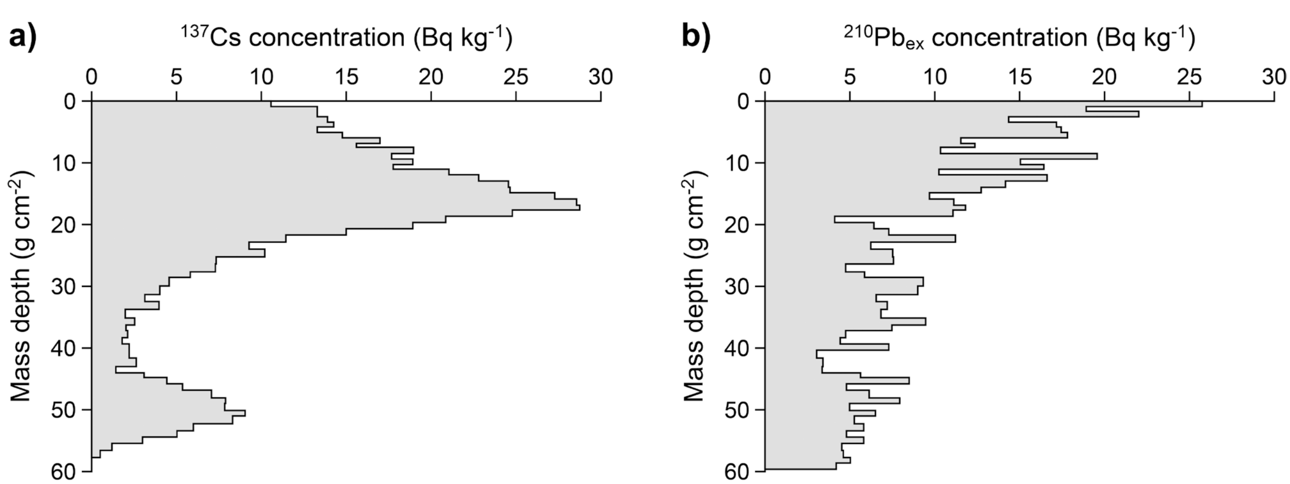

FIGURE 6. The ^{137}Cs (a) and ^{210}Pb$_{ex}$ (b) depth profiles associated with a sediment core collected from the floodplain of the River Severn at Welshpool, UK in 2009.

on the $^{210}Pb_{ex}$ measurements, and sedimentation rates of 1.38 $g \cdot cm^{-2} \cdot yr^{-1}$ for the period 1963–1986 and 0.70 $g \cdot cm^{-2} \cdot yr^{-1}$ for the period 1986–2009. It is beyond the scope of this contribution to speculate about the causes of these apparent changes in sedimentation rate, but such information clearly has considerable potential value for investigating the impact of human activity and particularly land use change, as well as climate change, on floodplain sedimentation and thus the sediment budgets of larger catchments.

CONCLUSIONS

The three examples outlined above serve to demonstrate the potential for up-scaling the use of fallout radionuclide measurements to the catchment scale in order to assemble the information on catchment sediment budgets needed to develop and implement effective sediment management strategies. The first approach described involved extrapolation. Here, alternative statistical models could be employed and scope undoubtedly exists to couple such models with geo-statistical techniques. In the second approach, emphasis was placed on collecting a relatively small dataset that was, nevertheless, representative of a particular catchment. In this case, further work is clearly needed to define the minimum number of samples required to characterize adequately a catchment and the optimum sampling framework for collecting those samples. The use of fallout radionuclides to document floodplain sedimentation rates was introduced to illustrate the need to consider additional components of the sediment budget, as the area of interest is up-scaled from a small to a larger catchment. Here also, further work is clearly required to explore and develop optimum strategies for moving from estimation of the sedimentation rate at a single point to providing an assessment of the mass of sediment sequestered within a given floodplain reach. This emphasises again the need to extrapolate a limited number of measurements of sedimentation rate or to assemble a limited dataset which can be viewed as representative of that reach.

Fallout radionuclide measurements have the potential to provide essentially unique information, which cannot be obtained using other techniques. By understanding the sources and sinks of sediment within a catchment, it is possible to implement sediment control measures aimed at reducing sediment fluxes in the most cost effective manner. However, important challenges undoubtedly remain to ensure that the full potential of using fallout radionuclide measurements to support catchment sediment management programmes is achieved.

ACKNOWLEDGEMENTS

The support of the FAO/IAEA Coordinated Research Programme D1.20.11, and associated Technical Contract 15478 for the work reported in this contribution is gratefully acknowledged. The work on the UK National Soil Erosion Inventory was funded by DEFRA contracts SP0411 and SP0413. Thanks are also extended to Jim Grapes for his help with gamma spectrometry measurements and to Helen Jones and Sue Rouillard for producing the figures.

REFERENCES

Belyaev, V.R., Golosov, V.N., Kislenko, K.S., Kuznetsova, J.S. & Markelov, M.V. 2008. Combining direct observations, modelling and ^{137}Cs tracer for evaluating individual event contribution to long-term sediment budgets. *In* J. Schmidt, T. Cochrane, C. Phillips, S. Elliott, T. Davies & L. Basher, eds. *Sediment dynamics in changing environments*. IAHS Publication no 325, pp. 114–122. Wallingford, UK, IAHS Press.

Du, P. & Walling, D.E. 2012. Using ^{210}Pb measurements to estimate sedimentation rates on river floodplains. *J. Environ. Radioact*., 103: 59–95.

Gellis, A.C. & Walling, D.E. 2011. Sediment source fingerprinting (tracing) and sediment budgets as tools in targeting river and watershed restoration programs. *In* A. Simon, S.J. Bennett & J.M. Castro, eds. *Stream restoration in dynamic fluvial systems: Scientific approaches, analyses, and tools*. Geophysical Monograph Series no. 194, pp. 263–291. Washington, DC. American Geophysical Union.

He, Q. & Walling, D.E. 2003. Testing distributed soil erosion and sediment delivery models using ^{137}Cs measurements. *Hydrol. Proc*., 17: 901–916.

IAEA. 1995. The use of fallout radionuclides in soil erosion investigations, by D.E.Walling & T.A. Quine. In *Nuclear techniques in soil-plant studies for sustainable agriculture and environmental preservation,* pp. 597–619. IAEA Publication ST1/PUB/947, Vienna.

Mabit, L., Bernard, C. & Laverdière, M.R. 2007. Assessment of erosion in the Boyer River watershed (Canada) using a GIS oriented sampling strategy and ^{137}Cs measurements. *Catena,* 71: 242–249.

Mabit, L., Benmansour, M. & Walling, D.E. 2008. Comparative advantages and limitations of the fallout radionuclides ^{137}Cs, $^{210}Pb_{ex}$ and ^{7}Be for assessing soil erosion and sedimentation. *J. Environ. Radioact.,* 99: 1799–1807.

Parsons, A.J. & Foster, I.D.L. 2011. What can we learn about soil erosion from the use of ^{137}Cs? *Earth-Sci. Rev.,* 108: 101-113.

Porto, P., Walling, D.E. & Callegari, G. 2011. Using ^{137}Cs measurements to estimate catchment sediment budgets and explore scale effects. *Hydrol. Process*., 25: 886–900.

Porto, P., Walling, D.E., Ferro, V. & Di Stefano, C. 2003a. Validating erosion rate estimates by caesium-137 measurements for two small forested catchments in Calabria, Southern Italy. *Land Degrad. Dev.,* 14: 389–408.

Porto, P., Walling, D.E., Tamburino, V. & Callegari, G. 2003b. Relating Cs-137 and soil loss from cultivated land. *Catena*, 53: 303–326.

Roehl, J.E. 1962. *Sediment source areas and delivery ratios, influencing morphological factors*. Proceedings Symposium on Land Erosion, Bari. IAHS Publication no. 59, pp. 202–213.Wallingford, UK, IAHS Press.

Walling, D.E. 1983. The sediment delivery problem. *J. Hydrol.,* 65: 209–237.

Walling, D.E. & Collins, A.L. 2008. The catchment sediment budget as a management tool. *Environ. Sci. Pol.,* 11: 136–143.

Walling, D.E. & He, Q. 1999. Improved models for estimating soil erosion rates from ^{137}Cs measurements. *J. Environ. Qual.,* 28: 611–622.

Walling, D.E. & Zhang, Y. 2004. Predicting slope-channel connectivity: A national scale approach. *In* V. Golosov, V. Belyaev & D.E. Walling, eds. *Sediment transfer through the fluvial system,* pp. 107–114. IAHS Publication no. 288, Wallingford, UK, IAHS Press.

Walling, D.E. & Zhang, Y. 2010. A national based assessment of soil erosion based on caesium-137 measurements. *Adv. Geoecol.,* 41: 89–97.

Zapata, F. ed. 2002. *Handbook for the assessment of soil erosion and sedimentation using environmental radionuclides*. Dordrecht, Kluwer. 221pp.

Use of an Integrated Approach for Assessing Soil Redistribution in the River Vorobzha Basin

V.N. Golosov[1,2,*] ,V.R. Belyaev[1], E.N. Shamshurina[1] and Ju.S. Kuznetsova[1]

ABSTRACT

Significant climate and crop rotation changes have taken place over the last few decades in the Central Chernozem zone of European Russia. These have considerably influenced sediment redistribution rates within agricultural catchments. Quantitative assessments of soil losses from cultivated lands and sediment redistribution intensity have been made using an integrated approach for typical catchments of Central Russia located near the city of Kursk in the River Vorobzha basin. Combined application of the caesium-137 (^{137}Cs) technique, erosion models, the soil morphological method and large-scale geomorphic mapping with detailed evaluation of the area of each typical morphological unit indicated that soil erosion rates on arable hill slopes decreased by about 1.5 times during the last 25 years (yr) compared with rates during the previous quarter century. Sediment delivery from cultivated fields into the River Vorobzha valley also decreased by up to three-fold over the same timeframes. The main reason for these changes was the considerably reduced surface runoff during the spring snowmelt period. In addition, the approach employed made it possible to quantify the effectiveness of soil conservation measures (SCMs) introduced within the small experimental sub-catchments occupying part of the study area. Application of SCMs reduced soil losses from cultivated fields by 2.5 times. Most of the eroded sediment and sediment-associated ^{137}Cs has been redeposited within the cultivated fields and adjacent dry valley bottoms. It can be concluded that taking an integrated approach is the most appropriate strategyfor assessing soil degradation and determining both localized and off-site soil and nutrient sinks.

Key words: *sediment redistribution, agricultural catchments, caesium-137, erosion models, soil conservation.*

INTRODUCTION

Soil loss from cultivated land causes land degradation and water pollution as a result of sediment and nutrient transport into river channels and water reservoirs. Quantitative assessment of sediment redistribution in different parts of fluvial systems can be made using an integrated approach to evaluate sediment budget components (Walling *et al.*, 2002; Polyakov *et al.*, 2004). Evaluation of sediment budgets provides a valuable basis for developing catchment management strategies through reliable identification of areas dominated by soil loss and locations of major sediment sinks. Caesium-137 (^{137}Cs) has been used successfully in different parts of the world as a tracer to construct sediment budgets as well as to assess mean annual erosion rates and the dynamics of deposition rates in areas of sediment storage (Mabit, Benmansour and Walling, 2008), including in European Russia (Golosov *et al.*, 2011). The southern part of European Russia is considered to be one of the most productive agricultural areas worldwide, in particular due to the high organic carbon content of its Chernozem soils. One of the most intensively cultivated areas of the Russian Chernozem zone is the Kursk region, characterized by extended areas of row crops such as sugar beet, potato and corn. Intensive cultivation started in this region in the 17th century, with the maximum percentage of cultivated land being reached after land tenure reform in 1861. Subsequently, the area of arable land decreased due to growth of urban areas and the cessation of tillage on the steepest slopes (Sidorchuk and Golosov, 2003). Intensive gully growth between 1861 and 1970 was a further reason for the abandonment of some cultivated parcels.

Soil conservation measures were first introduced at the beginning of the 1970s in the most eroded parts of the Russian Chernozem zone. Erosion associated with both spring snowmelt and summer rainstorms is responsible for soil degradation in this region. According to calculations based on the universal soil loss equation (USLE) erosion model, mean annual soil erosion rates in the Kursk region vary within the range 5 to 15 tonnes (t)$\cdot$ha$^{-1}\cdot$yr^{-1} (Litvin, 2002). Direct observations of snowmelt erosion indicated that the rate of the latter did not exceed 1–5 t$\cdot$ha$^{-1}\cdot$yr^{-1}. The highest erosion rates were observed during summer rainstorms, particularly on fields under fallow or row crops. For instance, in the Sovetsky district of the Kursk region, an extreme erosion event occurred on 20–21 August 1976 when about 190 mm of precipitation was recorded, causing soil losses of up to 200 t/ha during that single rainfall event (Gerasimenko and Rozhkov, 1976). More typical rainstorms with precipitation of about 20–40 mm can still cause soil losses of up to 36–44 t/ha for a single event when high intensity rainstorms fall over erosion-prone surfaces such as harrowed fallow or row crops (Belyaev *et al.,* 2008).

The main aim of the present study was to evaluate the dynamics of erosion and sediment redistribution rates for typical catchments of different sizes located in the Chernozem zone (the Kursk region) using an integrated approach. Its importance is underlined by the fact that during the last decades considerable climatic fluctuations in the study area coincided with notable crop rotation changes. In addition, this study illustrates the applicability of the integrated approach used for quantitative assessment of the effectiveness of soil conservation measures (SCMs).

1 Laboratory for Soil Erosion and Fluvial Processes, Faculty of Geography, Lomonosov Moscow State University, Leninskie Gory, 1, 119991, Moscow, Russia
2 Institute of Ecology and Geography, Kazan (Volga Region) Federal University, Kremlevskaya, 18, Kazan, 420008, Russia
* E-mail address of corresponding author: gollossov@gmail.com

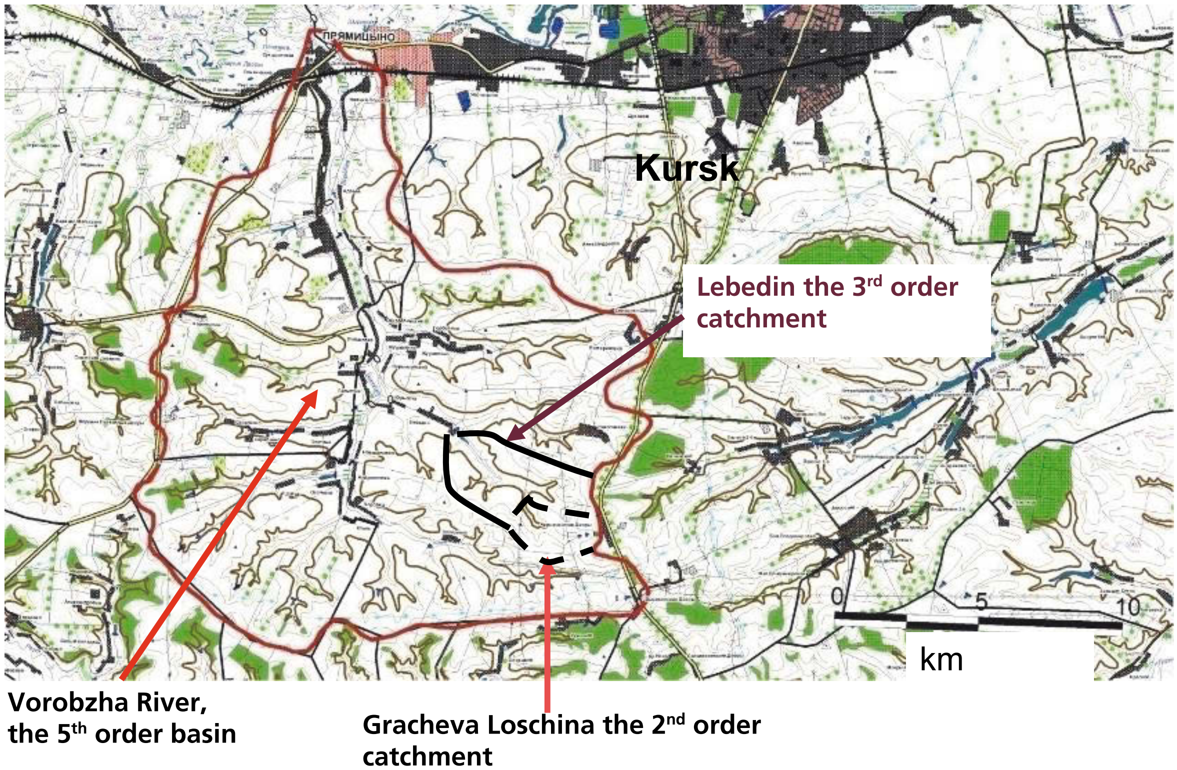

FIGURE 1. Location of the study catchments.

STUDY SITE

The River Vorobzha basin is a typical small river basin for the agricultural area of the Chernozem zone of Central Russia. It is located in the Kursk region 15–20 km SSW from the regional centre Kursk. It has an area of 228 km^2 with about 85 percent of cultivated land. The territory is characterized by a continental climate with relatively cold winters and hot summers. Average annual precipitation is 585 mm (over a 100–yr period of observation), varying from 400–800 mm. About 30 percent of precipitation falls during the cold months mostly as snow. Rainstorms with total precipitation of 10–40 mm and occurring commonly from May to October are the most typical precipitation events. The River Vorobzha basin was contaminated by radionuclide fallout after the Chernobyl accident in 1986. The level of Chernobyl-associated contamination is about ten times higher than the bomb-derived ^{137}Cs contamination which took place between 1954 and the 1970s.

Assessments of soil redistribution rates were undertaken for the Gracheva Loschina catchment (area 1.98 km^2) and the Lebedin catchment (area 15.2 km^2), which are typical 1st and 3rd Hortonian order catchments within the River Vorobzha basin. Both catchments have earthen dams at their outlets which were constructed in 1986 and 1956, respectively. These enabled calculation of detailed sediment budgets for both catchments. In addition, experimental SCMs have been employed on 70 percent of the Gracheva Loschina catchment since the beginning of 1986. Hence it was also possible to evaluate the impact of the introduction of SCMs on soil erosion and sediment redistribution.

METHODS

An integrated approach (involving erosion model calculations, ^{137}Cs budget, and sediment sequence dating, and large-scale geomorphic mapping) was used for quantitative assessment of soil loss/gain on arable hillslopes and in different order catchments over several time periods since the middle of the 20th century. In addition, detailed information about land use and crop rotation changes and meteorological data over a 100 yr period was collected for the study area. Principles of the integrated approach are based on applying at least two independent methods to assess each component of a sediment budget and the possibility of extrapolating the results of more detailed evaluation of soil redistribution obtained for relatively small catchments in order to cover larger areas (Belyaev *et al.*, 2009; Golosov *et al.*, 2011).

During the first stage of the study, large-scale geomorphic mapping was used for area evaluation of different morphological units within the Gracheva Loschina catchment. Mean soil loss/gain within each morphological unit was determined using ^{137}Cs techniques, the soil morphological method (Kiryukhina and Serkova, 2000) and erosion model calculations. As a result, it was possible to check the correctness of each technique applied. These techniques enabled calculation of the Gracheva Loschina catchment sediment budget based on evaluation of soil losses from arable hillslopes, sediment redeposition within cultivated slopes and aggradation of uncultivated valley banks and dry valley bottoms, including deposition in small reservoir upstream from the earthen dam at the catchment outlet. Values for soil redistribution rates and sediment delivery ratios determined for different geomorphic units within a smaller catchment were then used to determine sediment budgets for larger

TABLE 1. The post-1986 ^{137}Cs budget for sub-catchments of the Gracheva Loschina catchment characterized by different types of soil conservation measures or their absence

Sub-catchment	Total area (ha)	Eroded area ^{137}Cs loss, kBq	Deposition area, ^{137}Cs gain, kBq (% from ^{137}Cs loss)		Residual kBq (%)
			Within cultivated areas (including grassed waterways)	In tributary hollow bottoms and main valley bottom	
Sub-catchment with forest shelter belts and grass waterways	52.8	189 606 (100%)	154 620 (82%)	7 954 (4%)	27 032 (14%)
Sub-catchment with forest shelter belts, grass waterways and contour terraces	88.1	926 885 (100%)	834 195 (90%)	24 650 (3%)	68 040 (7%)
Area without soil conservation measures and the main valley bottom	56.9	236 786 (100%)	22 061 (9%)	200 508 (85%)	14 217 (6%)

catchments based on distinguishing similar types of morphological units and estimating erosion rates by the erosion model. The resulting values for sediment delivery into the main valley of the Lebedin catchment were tested by comparison with sedimentation rates and volumes obtained from analysis of the ^{137}Cs-based valley bottom sediment microstratigraphy, including the dry reservoir infill.

RESULTS AND DISCUSSION

Analysis of bomb-derived ^{137}Cs inventories at different geomorphic landscape units can be applied effectively for evaluating sediment budgets in small catchments (Loughran *et al.*, 1992; Owens *et al.*, 1997; Walling *et al.*, 2002). The authors attempted to apply this approach for the Gracheva Loschina catchment and to compare the results with those obtained by other methods.

The catchment topography is characterized by gently rolling interfluve areas and a predominance of convex slopes with maximum gradients of between 5 and 8 degrees. Several hollows dissect the slopes in the upper part of the catchment. Before 1986, most of the catchment area was cultivated. In 1986, the lower parts of the slopes and the hollow bottoms of the tributary were sown with grass and converted into pasture. In addition, SCMs were introduced on 70 percent of the catchment area (sub-catchments of the two upper tributaries) in March 1986 as part of soil conservation field experiments. Two-rowed forest shelter belts were planted parallel to the slope contours at a spacing of about 200–250 m, and grassed waterways were constructed along the bottoms of the hollows within both experimental sub-catchments. Water retention ditches with a depth of about 1 m were dug within each forest shelter belt between the two rows of trees. The bottoms of hollows were sown with perennial grasses, left continuously uncultivated and used as erosion-protected flow pathways for concentrated surface runoff. In addition, contour terraces parallel to the contour lines with a relative height of about 1 m were constructed between the forest shelter belts within one of the sub-catchments with a spacing of about 50 m between the adjacent terraces. Simultaneously, a closed earthen dam was constructed at the main valley outlet. The rest of the catchment slopes downstream from the main tributary confluence remained cultivated in the traditional manner.

The Chernobyl fallout occurred at the end of April 1986, shortly after the introduction of the SCMs. The Gracheva Lochina catchment therefore prov ided a unique opportunity to establish a closed system sediment budget based on the redistribution of the Chernobyl-derived ^{137}Cs, and to test the reliability of the results by comparison with independent approaches. In order to design a representative sampling programme, a detailed large-scale geomorphic map was created based on a combination of already available topographic data (1:10 000 scale map with 1 m contour intervals), and additional differential global positioning system (DGPS) and digital tacheometer surveys carried out in selected parts of the catchment. The sampling programme aimed to characterize all important geomorphic units in terms of ^{137}Cs inventory and, subsequently, sediment redistribution between them. The area of each geomorphic unit was determined based on the geomorphic map constructed. The ^{137}Cs budget was compiled for each of the three sub-catchments distinguished on the basis of different post-1986 land use patterns (Table 1).

Uncertainty with respect to the calculations (represented by the *Residual* column) arises from the presence of bomb-derived ^{137}Cs and redistribution prior to the introduction of SCMs were not accounted in the budget calculations. According to available information (Izrael, 1998), the ratio between the Chernobyl-derived and bomb-derived ^{137}Cs inventories (corrected for radioactive decay) for the case study area is about 6:1–5:1. It is therefore suggested that part of the bomb-derived ^{137}Cs inventory would have been removed through the catchment outlet before construction of the dam in 1986. This can explain some of the differences between ^{137}Cs losses from eroded areas and ^{137}Cs accumulation within deposition areas. However, it is also clear from Table 1 that most of the Chernobyl-derived ^{137}Cs was delivered into the main valley bottom from areas with traditional cultivation. For the two sub-catchments with SCMs the situation is the opposite — more than 95 percent of Chernobyl-derived ^{137}Cs has remained within the slopes. This emphasizes the marked differences in sediment delivery into the main valley between sub-catchments with different land uses. Mechanical soil translocation towards contour terraces during tillage operations in the experimental area can explain the ^{137}Cs redistribution observed within the cultivated area of the northeastern sub-catchment with the SCM applied. In the southeastern SCM sub-catchment with only forest shelter belts, most of the ^{137}Cs redistribution within the slopes can be attributed to the limited erosion and subsequent sediment redeposition within grassed waterways.

The sediment budget was calculated from the ^{137}Cs total inventory data using the following equation based on the simple proportional calibration model:

$$R = \frac{\int_S A\mathrm{d}S - \int_S A_{\mathrm{ref}}\mathrm{d}S}{C_p \Delta t S}$$

where R is mean annual soil loss/gain, $kg \cdot m^{-2} \cdot yr^{-1}$ (negative values mean erosion, positive mean accumulation); Δt is time elapsed since fallout of Chernobyl-derived ^{137}Cs, yr; C_p is ^{137}Cs concentration in

TABLE 2. Post-1986 sediment redistribution (expressed as t and percent (in brackets)) within the Gracheva Loschina catchment calculated by two different approaches

Method	Time interval (years)	Gross erosion (t/%)	Deposition within cultivated field (t/%)	Deposition within hollows and valley bottom (t/%)	Output from catchment (t/%)
^{137}Cs budget	1986–2006	50 989 (100)	33 778 (82.8)	8766 (17.2)	0 (0)
Erosion model calculation and sediment deposition in the valley bottom (vertical distribution of ^{137}Cs)	1986–2006	22 606 (100)	17 050 (75.4)	5556 (24.6)	0 (0)

TABLE 3. Mean annual erosion rates and total soil losses from cultivated slopes of the Lebedin catchment for three time intervals covering the entire period after dam construction in catchment outlet

Time interval (years)	Mean annual erosion rate ($t \cdot ha^{-1} \cdot yr^{-1}$)	Total soil losses (t)
1956–1964	8.5	83 912
1964–1986	10.6	287 769
1986–2008	6.8	191 848
Mean for 1956–2008	8.6	563 529

plough layer, Bq/kg; A ius ^{137}Cs inventory at sampling point, Bq/m; $\int_S A_{ref} dS$ is total ^{137}Cs fallout inventory within a geomorphic unit, Bq; $\int_S A dS$ is total ^{137}Cs inventory within geomorphic unit, Bq; and S is area of geomorphic unit, m^2.

Data for individual geomorphic units were integrated to provide the sediment budget for the entire catchment over the post-1986 period. Table 2 indicates that according to the sediment budget derived from the ^{137}Cs budget, more than 80 percent of the sediment eroded from arable hillslopes was redeposited within the slopes after introduction of SCMs, while only <20 percent was delivered to valley bottoms.

The higher values of gross erosion and within-slope redeposition provided by the ^{137}Cs budget approach can be explained, using the case study with individual soil transects. In fact the ^{137}Cs technique gives an integral evaluation of soil redistribution including both water erosion and tillage translocation. The latter is especially important in the sub-catchment with contour terraces.

It can be concluded that the application of Chernobyl-derived ^{137}Cs for establishing the sediment budget of a small cultivated catchment produced results comparable with data obtained by independent techniques. In addition, it allows the effectiveness of various SCMs to be evaluated through comparisons with the sediment budgets constructed for earlier periods using methods with different temporal resolutions (Golosov *et al.*, 2008). Application of SCMs in the Gracheva Loschina catchment since 1986 has reduced average soil loss rates by at least a factor of 2.5 based on the ^{137}Cs budget method.

After dam construction in 1956 only about 10 percent of sediment was transported outside the Lebedin catchment (Prytkova, 1981). Gross erosion rates for arable hillslopes were calculated for known cultivated areas and crop rotations using the USLE-based model for the periods 1956–1964, 1964–1986 and 1986–2008 and are provided in Table 3.

Analysis of the vertical distribution of ^{137}Cs at several incremental depths and depositional locations enabled evaluation of deposition rates over different time intervals. The most intensive deposition (about 2 cm per yr for the post-Chernobyl period) occurred in infills of secondary discontinuous valley bottom gullies. However, such valley bottom cuts are typically only about 2–3 m wide. Deposition rates on other parts of dry valley bottoms were within the range of 0.8 to 1.0 cm/yr, while for the 1964–1986 period, rates ranged between-1.0 and 1.2 cm/yr. Analysis of ^{137}Cs data from the sediment section located in the dried reservoir suggest that before 1986 the intensity of sediment transport during the concentrated snowmelt runoff period was much higher than that during the post-Chernobyl period. This conclusion was based on the location of the Chernobyl-associated peak of ^{137}Cs distribution at a depth about 7 cm below the surface. On the other hand, the deposition layer for the 1956–1986 period had a thickness of about 68 cm. Hence, the intensity of sediment delivery to the Lebedin catchment outlet was about 10 times higher during the latter period. Nevertheless, it is necessary to take into consideration the fact that during reservoir infill by sediment, the gradient of the valley bottom decreased considerably, promoting more active sediment redeposition at the upper part of the reservoir. These observations are consistent with the results of monitoring data on spring snowmelt runoff on cultivated slopes at the Novosil experimental station (Orel region) that demonstrated a considerable decrease in runoff coefficients during the last 12 years (Petelko, Golosov and Belyaev, 2007). A similar trend was found during monitoring observations in the upper part of the Lebedin catchment (the Gracheva Loschina experimental catchment) for the 1986–2002 period (Kumani, 2003).

Detailed determination of areas consisting of different morphological units (including main bottom level, 1–2 terrace levels and discontinuous valley bottom gullies) within the main valley bottom (3rd Hortonian order) of the Lebedin catchment and its tributaries of 1st and 2nd Hortonian orders was carried using tacheometric and global positioning system (GPS) surveys. The total volume of sediment deposited was calculated based on morphological unit areas and ^{137}Cs-based aggradation rates for the main bottom of the Lebedin catchment valley (3rd order) and Gracheva Loschina catchment valleys (1st and 2nd order valleys) (Table 4).

In addition to sediment deposition on the valley bottoms, other sediment sink zones exist. These include: redeposition within cultivated fields; redeposition along the lower boundaries of cultivated fields (at plough terraces); steep grassed dry valley sides; and uncultivated parts of slope hollows (infilled valley side gullies). It is difficult to calculate the sedimentation volume for each of these zones accurately. However, it is possible to estimate the percentage of sedimentation in each zone based on observation data during extreme erosion events and detailed measurements undertaken within the Gracheva Loschina catchment. Redeposition within the cultivated fields based on direct measurements after snowmelt and rainstorm runoff events usually varies within a range of 2 to 25 percent of gross soil values for the respective fields. It is also confirmed by detailed evaluation of gross and net erosion rates at slope catchments using the ^{137}Cs technique (Golosov *et al.*, 2011). Average within-field sediment rede-

TABLE 4. Total sediment deposition in valley bottoms of different orders (Lebedin catchment) for different time intervals

Valley	Hortonian order of the valley	Sediment deposition (t)		
		1956–1964	1964–1986	1986–2008
Gracheva Loschina	1	1 260	5 044	250
	2	2 814	11 256	10 553
Other valleys	1	5 591	22 365	16 773
	2	12 665	50 661	39 579
	3	7 156	28 625	28 625
Total volume in valley bottoms upstream from reservoir		29 486	117 951	95 555
Reservoir		46 769		
Total volume		289 761		

TABLE 5. Percentage of sediment re-deposited in different deposition zones of the Lebedin catchment for the 1956–2008 period

Deposition Zone	% sediment re-deposited in given zone (from gross soil losses)	
	Range (based on observation)	Mean value
Within arable lands	2–25	10
Near the lower edge of cultivated fields	5–30	8
Valley banks	2–7	4
Uncultivated part of hollows (former bank gullies)	4–10	6
Bottom of valleys including pond	—	51
Beyond the pond	5–15	10
Residual	—	11

TABLE 6. Sediment redistribution within the Lebedin catchment estimated for different time intervals based on evaluation of sediment deposition in valley bottoms and corrected results of soil loss calculations by empirical models

Time interval	Total gross soil losses (t) (based on empirical model calculations)	Sediment volume in the valley bottoms (t) (based on ^{137}Cs dating)	Hillslope-mobilized sediment redeposited in the valley bottoms (%)	Hillslope-mobilized sediment redeposited along pathways from cultivated slope to valley bottoms (%)
1956–1964	83 912/74 682*	37 858	45/51*	39
1964–1986	28 7769/256 114	151 438	53/59	31
1986–2008	191 848/170 745	100 465	52/59	31
1956–2008	563 529/506 881	289 761	51/57	33

* numerator — results of soil losses calculation without correction; denominator — results of soil losses calculation with correction based of sediment budget calculations

position is around 10 percent of total eroded volume. Acummulation along the lower boundaries of cultivated fields (at plough terraces) is usually more significant and varies between 5 percent and 30 percent of gross soil losses according to different measurements and observations. For the Gracheva Loschina catchment, it was determined that 8 percent of hillslope-mobilized material remained stored in plough terraces.

The most difficult task is to estimate sediment deposition on the grassed valley sides because of the extremely random nature of this process. Data obtained after extreme erosion events and during snowmelt periods showed that between 2 and 7 percent of sediment mobilized from cultivated land was redeposited within these morphological units (Belyaev *et al.*, 2008). Sediment deposition in uncultivated parts of slope hollows (most of which actually represents infilled formerly active valley side gullies) was calculated based on direct measurements of total sediment volumes stored in hollows of the Gracheva Loschina catchment and the total number of uncultivated hollows on the valley sides of the Lebedin catchment (Table 5). By combining all the above data and comparing the results with the USLE-based soil loss calculations, it was concluded that the empirical erosion model overestimated total soil loss from the cultivated slopes of the Lebedin catchment by about 11 percent.

Comparision of total soil losses from cultivated hillslopes with sediment deposition in the valley bottoms also demonstrated that the proportion of sediment deposited in valley bottoms from total soil losses did not differ significantly between the different time intervals considered (Table 6). It can therefore be suggested that empirical models consistently over-estimate soil losses and that it is very likely that actual soil losses are 11 percent lower than those calculated. Based on this assumption, 57 percent of material eroded from cultivated slopes remained redeposited in the valley bottoms, 33 percent along pathways from cultivated slopes to the valley bottoms, and only about 10 percent of sediment was transported further downstream and reached the catchment outlet reservoir (Table 6).

TABLE 7. Proportion of sediment potentially exported from the Lebedin catchment into the River Vorobzha valley for different time intervals (if the catchment outlet reservoir did not exist)

Time interval	Total gross soil losses (t) (based on corrected empirical model calculations)	Sediment volume in valley bottoms (t) (based on ^{137}Cs dating and area of bottoms)	Sediment deposited in pond (t)	Percentage of sediment potentially exported from the Lebedin catchment (%)
1956–1986	330 796	189 296	39 754	12/21*
1986–2008	170 745	100 465	7 015	4/7

*numerator — % from total soil losses on cultivated hillslopes; denominator — % from deposition in valley bottoms

It can be suggested therefore, that the volume of sediment redeposited within the main reservoir at the catchment outlet was proportional to the volume of sediment potentially exported from the Lebedin catchment. In this respect, it is noteworthy that before 1986 about 12 percent of total soil losses or about 21 percent of total deposition in the valley bottoms reached the catchment outlet. After 1986 the percentage of sediment reaching the Lebedin catchment outlet decreased by more than threefold (Table 7), mainly as a result of lower surface runoff and erosion during the spring snowmelt period. Consequently, it can be tentatively suggested that during the last two decades, the volume of sediment delivered into the River Vorobzha valley from its main tributary catchments fell by about one third.

CONCLUSIONS

This study shows that changes in climate and crop rotation reduced significantly the rates of soil loss from cultivated fields within typical catchments of the Chernozem zone of Central European Russia during the last two decades. Most of the eroded sediment and sediment-associated ^{137}Cs were redeposited within cultivated fields and dry valley bottoms. Application of an integrated approach involving several independent techniques allowed detailed and reliable evaluation of soil degradation rates and the contributions of both local and off-site soil sinks.

ACKNOWLEDGEMENTS

This research work was carried out with the financial support of the Russian Foundation for Basic Research (RFBR grants no. 07-05-00193 and 10-05-00976), the International Atomic Energy Agency Research Contract no RUS/15482, and the President of the Russian Federation Support Programme for young researchers (Project nos. MK-8023.2010.5 and MK-1221.2012.5) and for leading scientific schools (Project no. NS-79.2012.5).

REFERENCES

Belyaev, V.R., Golosov, V.N., Kislenko, K.S., Kuznetsova, J.S. & Markelov, M.V. 2008. Combining direct observations, modelling and ^{137}Cs tracer for evaluating individual event contribution to long-term sediment budgets. *In* J. Schmidt, T. Cochrane, C. Phillips, S. Elliott, T. Davies & L. Basher, eds. *Sediment dynamics in changing environments*. IAHS Publication no 325, pp. 114–122. Wallingford, UK, IAHS Press.

Belyaev, V.R., Golosov, V.N., Kuznetsova, J.S. & Markelov, M.V. 2009. Quantitative assessment of effectiveness of soil conservation measures using a combination of ^{137}Cs radioactive tracer and conventional techniques. *Catena,* 79: 214–227.

Gerasimenko, V.P. & Rozkov, A.G. 1976. Remarkable rainstorm in Central Chernozem Region and display of erosion processes. *Zaschita pochv of erozii,* 4: 6–21.

Golosov, V.N., Belyaev, V.R., Kuznetsova, Yu.S., Markelov, M.V. & Shamshurina, E.N. 2008. Response of a small arable catchment sediment budget to introduction of soil conservation measures. *In* J. Schmidt, T. Cochrane, C. Phillips, S. Elliott, T. Davies & L. Basher, eds. *Sediment dynamics in changing environments*. IAHS Publication no 325, pp. 106–113. Wallingford, UK, IAHS Press.

Golosov, V.N., Gennadiev, A.N., Olson, K.P., Markelov, M.V., Zhidkin., A.P., Cendev, Yu.G. & Kovach, R.G. 2011. Spatial and temporal features of soil erosion in the forest-steppe zone of the East-European Plain. *Eurasian Soil Sci.,* 7: 794–801.

Kiryukhina, Z.P. & Serkova, Yu.V. 2000. Podsolic soils morphometric characteristics variability and the diagnostics of soil erosion. *Soil Erosion Channel Process.,* 12: 63–70.

Kumani, M.V. 2003. *Methods of regulation of soil erosion processes and hydrological regime of agrolandscapes of Central Chernozem zones*. University of Kursk, (DSc thesis).

IAEA. 2011 Application of radionuclide technique and other methods for assessing the effectiveness of soil conservation measures at Novosil study site, Orel region by V.N. Golosov, V.R. Belyaev, M.V. Markelov, N.N. Ivanova & Y.S. Kuznetsova. In *Impact of soil conservation measures on erosion control and soil quality,* pp. 131–157. IAEA-TECDOC 1665, Vienna.

Izrael, A.Yu. (ed). 1998. *Atlas of radioactive contamination of European Russia, Belarus and Ukraine.* Moscow, Russia, Federal Service of Geodesy and Cartography of Russia. 143 pp.

Litvin, L.F. 2002. *Geography of soil erosion on agricultural lands of Russia*. Moscow, "Akademkniga". 256 pp.

Loughran, R.J., Campbell, B.L., Shelly, D.J. & Elliot, G.L. 1992. Developing a sediment budget for a small drainage basin in Australia. *Hydrol. Process.,* 6: 145–158.

Mabit, L., Benmansour, M. & Walling, D.E. 2008. Comparative advantages and limitations of fallout radionuclides (^{137}Cs, ^{210}Pb and ^{7}Be) to assess soil erosion and sedimentation. *J. Environ. Radioact.,* 99: 1799–1807.

Owens, P.N., Walling, D.E., He, Q., Shanahan, J. & Foster, I.D.L. 1997. The use of caesium-^{137} measurements to establish a sediment budget for the Start catchment, Devon, UK. *Hydrol. Sci. J.,* 42: 405–423.

Petelko, A.I., Golosov, V.N. & Belyaev, V.R. 2007. *Experience of design of system of counter-erosion measures*. Proceedings 10th International Symposium on River Sedimentation Vol. 1, pp. 311–316. Moscow, Russia.

Polyakov, V.O., Nearing, M.A. & Shipitalo, M.J. 2004. Tracking sediment redistribution in a small watershed: Implication for agro-landscape evolution. *Earth Surf. Process. Landforms*, 29: 1275–1291.

Prytkova, M.Y. 1981. *Sedimentation in small reservoirs*. Leningrad, Nauka. 180 pp.

Sidorchuk, A.Yu. & Golosov, V.N. 2003. Erosion and sedimentation on the Russian Plain, II: The history of erosion and sedimentation during the period of intensive agriculture. *Hydrol. Process.*, 17: 3347–3358.

Walling, D.E., Russell, M.A., Hodgkinson, R.A. & Zhang, Y. 2002. Establishing sediment budgets for two small lowland agricultural catchments in the UK. *Catena,* 47: 323–353.

Assessment of Soil Erosion and Sedimentation Rates in 'My Bouchta' Watershed in North Morocco using Fallout Radionuclides and Stable Isotopes

M. Benmansour[1,*], N. Amenzou[1], A. Zouagui[1], H. Marah[1], M. Sabir[2], A. Nouira[1], H. Benjelloun[2], A. Benkdad[1] and F. Taous[1]

ABSTRACT

The aim of this work was to combine measurements of fallout radionuclides (FRN's) Caesium-137 (^{137}Cs) and excess Lead-210 ($^{210}Pb_{ex}$) with stable isotopes of carbon and nitrogen ($\delta^{13}C$ and $\delta^{15}N$) for investigating land degradation and the origin of sediment deposits in the My Bouchta watershed in northern Morocco. FRN's were used as tools for obtaining quantitative information on soil erosion and sedimentation rates over a range of different timescales whilst the stable isotopes enabled the primary sediment source areas to be identified. Using ^{137}Cs the net soil erosion rate for the My Bouchta watershed was estimated to be about 22.1 tonnes (t)·ha^{-1}·year^{-1} over a period of 50 years with a major contribution from agricultural fields. Net soil erosion rates over a period of 100 years derived from $^{210}Pb_{ex}$ were lower than those estimated by ^{137}Cs. These results indicate that soil erosion has increased significantly during the last 50 years. Sedimentation rates of about 0.50 g·cm^{-2}·year^{-1} (equivalent to 50 t·ha^{-1}·year^{-1}) were obtained for the Talembout water reservoir suggesting additional contributions from gully erosion and mass movement. Similar behaviours of stable isotopes with depth were obtained for forest and shrub fields and a good correlation between $\delta^{13}C$ and total C was obtained for forest and shrub fields. Using the $\delta^{13}C$ profile, sediment deposits in the water reservoir seemed to originate mainly from fields under maize culture (C_4 plant). The sediment profile of $\delta^{15}N$ indicated an increased use of synthetic fertilizers during the last 15 years.

Key words: *fallout radionuclides, carbon, nitrogen, stable isotopes, soil erosion, sedimentation.*

INTRODUCTION

Land degradation by soil erosion is a major concern for Moroccan watersheds especially in the north, leading to considerable on-site and off-site impacts. As reported by Namr and Mrabet (2004), the main on-site impact on the Moroccan agricultural landscape is reduced soil fertility while the main off-site impact is increased siltation in water reservoirs. Out of 22.7 million ha of potentially exploitable land in the northern part of Morocco, 77 percent is estimated to be subject to very high erosion risks (Belkhiri, 1988). However, few reliable datasets are available on the extent of land degradation in many Moroccan basins. Integrated data are therefore needed to develop soil conservation strategies and protect water reservoirs against sedimentation and pollution at the watershed scale. Most previous studies have used classical techniques such as experimental plots or prediction models to estimate soil erosion and its impacts. Although nuclear techniques have been used in Morocco for some years (e.g. Bouhlassa, Moukhchane and Aiachi, 2000; Nouira, Sayouty and Benmansour, 2003; IAEA, 2011), their application to study events at the watershed scale is limited. This paper presents preliminary results of a study which used an integrated approach involving FRNs (Walling, He and Quine, 1995; Zapata 2002; Mabit, Benmansour and Walling, 2008) and stable isotopes (Phillips and Greeg, 2002; Gibbs, 2008) to establish a complete sediment budget and identify soil erosion sources at the My Bouchta watershed.

MATERIALS AND METHODS

The watershed My Bouchta is located in the occidental part of the Rif mountains between the cities of Tetouan and Chaouen (Figure 1). The total area is 76.6 km^2 and the mean annual precipitation is about 800 mm. The watershed is characterized by high slopes (from 20 to 40 percent), and the soils are degraded by different forms of very active water erosion: sheet, rill, gully erosion and mass movement. Based on the information collected and preliminary visits to the study site, a sampling strategy was established for the whole watershed, including upland to lowland areas using a geographic information system tool (ArcGIS). It was based on selecting representative fields taking into account the different classes of land use, topography and soil type. The main classes selected for these three parameters were as follows:

- Three classes for the land use: forest (FR), shrub (SR) and agricultural (AG) areas.
- Four classes for the slope gradient (in percent): from 0 to5, from 5 to15, from 15 to 30 and > 30.
- Four classes for the soil type: vertisols (VE), alfisols (AL), aridisols (AR) and entisols (EN).

All together 48 homogenous units or iso-sectors were defined. To optimise the sampling strategy, only 22 important units covering

1 Centre National de l'Energie, des Sciences et des Techniques Nucléaires (CNESTEN), BP.1382, R.P 10001, Rabat, Morocco
2 Ecole Nationale Forestière d'Ingénieurs (ENFI), B.P. 511, Salé, Morocco
* E-mail address of corresponding author: benmansour64@yahoo.fr

L.K. Heng, K. Sakadevan, G. Dercon and M.L. Nguyen (eds), Proceedings — International Symposium on Managing Soils for Food Security and Climate Change Adaptation and Mitigation. Food and Agriculture Organization of the United Nations, Rome, 2014: 113–118

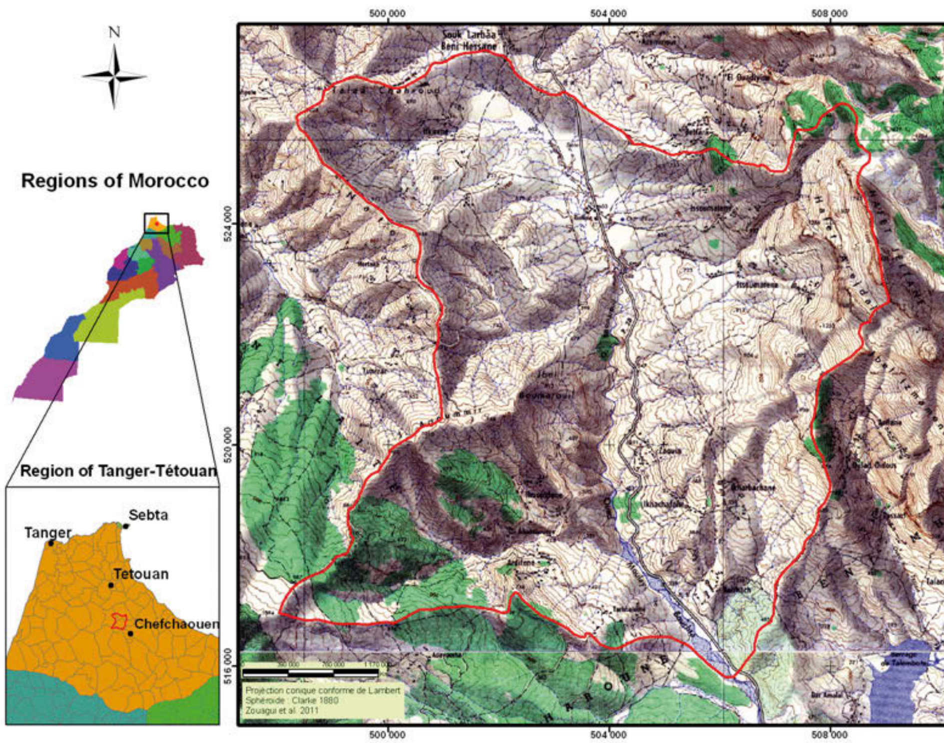

FIGURE 1. Location of My Bouchta watershed.

more than 90 percent of the total watershed area were considered for the study. A transect approach (from 1 to 4 transects) was adopted to collect samples from the top to the bottom of the field. Until now, around 200 soil and sediment samples were collected in various fields representing ten homogenous sectors in upland areas of the watershed, four reference sites (RF1, RF2, RF3 and RF4) and one area of the Talembout water reservoir (WR1, WR2, WR3). A motorized cylindrical tube (diameter ca. 9 cm) or the so-called Column Cylinder Auger was inserted to a depth of between 30 and 40 cm to ensure that all ^{137}Cs or ^{210}Pb were measured. Regarding the use of the stable isotopes ($\delta^{13}C$ and $\delta^{15}N$), the transect approach was also used from the top to the bottom, and soil profiles were taken with the same core sampler at six sampling points per transect (distance between sampling points was approximately 20 m). At each sampling point a soil core was taken and divided into four subsamples representing incremental depths (from 0 to10, 10 to 20, 20 to30 and 30 to 40 cm).

The soil and sediment samples taken for radionuclide analyses were dried, lightly ground, sieved (<2 mm) and homogenised prior to the measurement of ^{137}Cs, ^{210}Pb, ^{226}Ra (from ^{214}Bi) by gamma spectrometry using an HPGe detector (Canberra p-type 30 percent). Calibration of the detection systems was done by preparing standards from a certified multi-gamma source (Amersham) and IAEA reference materials (IAEA 327 and 375). Generally, samples were placed in Marinelli bakers (500 ml) or cylindrical containers (200 ml). ^{137}Cs, ^{210}Pb and ^{214}Bi activities were determined from the net gamma ray peak areas at 662, 478 and 46.5 keV, respectively. Counting rates varied from 12 to 24 h which provided a precision of about 5 percent to 20 percent at 95 percent level of confidence. Indirect determination of excess ^{210}Pb activity from total ^{210}Pb and ^{226}Ra gave a low precision for activity values (range 30 percent to 50 percent). To improve the precision of ^{210}Pb activity, measurements by alpha spectrometry through ^{210}Po (daughter of ^{210}Pb), was performed for some soil and sediment samples with low activities. This method required a total digestion of soil samples containing ^{210}Pb, ^{210}Po and tracer ^{209}Po, using HNO_3, HCl and HF acids and spontaneous deposition of polonium isotopes in silver discs.

Counting was carried out using silicon detectors (EG & G, Ortec). Alpha spectrometry provides a good precision for measuring ^{210}Pb but rquires longer counting times than gamma spectrometry. Quality control procedures used control charts (efficiency, resolution and background), certified reference materials and regular participation in inter-comparison exercises and proficiency tests organised by the IAEA.

For the stable isotopes, as even small amounts of inorganic C can lead to significant errors in $\delta^{13}C$, inorganic C was removed by acid fumigation prior to analysis. The samples were stored at approximately 4°C until further processing. Moistened sub-samples were exposed overnight to HCl. Afterwards the samples were rinsed by shaking with 2 ml of distilled water, dried at 40°C and ground in a mortar before measuring the stable isotope ratio. Stable C isotope analysis was carried out using a continuous flow isotope ratio mass spectrometer (Delta Plus) coupled with an elemental analyzer (Carlo Elba, NC2500). Carbon and N stable isotope abundances were expressed as follows as delta (δ) values to indicate differences between the isotopic ratio of the sample and accepted standard materials:

$$\delta X(‰) = \left(\frac{R_{sample}}{R_{std}} - 1\right) \times 10^{3}$$

where: X — (^{13}C or ^{15}N), R_{sample} — the isotope ratio (^{13}C:^{12}C or ^{15}N:^{14}N) of the sample; and R_{std} = the isotope ratio of the standard Vienna Pee Dee Belemnite (VPDB), and atmospheric N, respectively.

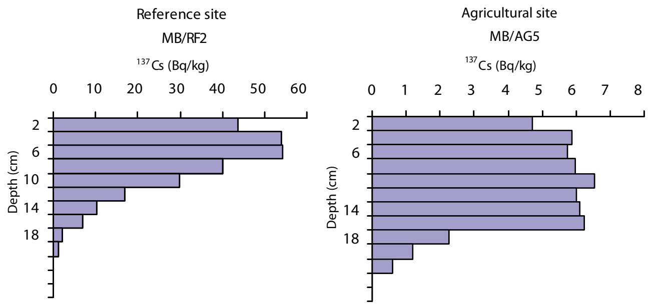

FIGURE 2. Example of ^{137}C depth profiles associated with a reference and an agricultural site at 'My Bouchta' watershed.

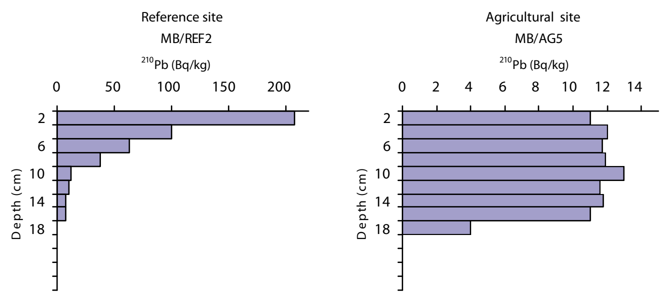

FIGURE 3. Example of $^{210}Pb_{ex}$ depth profiles associated with a reference and an agricultural site at My Bouchta watershed.

RESULTS AND DISCUSSION

Soil erosion and sedimentation rates using ^{137}Cs and $^{210}Pb_{ex}$

Typical depth profiles of ^{137}Cs and $^{210}Pb_{ex}$ activities associated with the reference (forest with no slope) and cultivated sites are shown in Figures 2 and 3. Most ^{137}Cs and $^{210}Pb_{ex}$ were contained in the top 10 cm and, as expected, concentrations decreased exponentially with depth. For the cultivated site, concentrations were almost uniform throughout the plough layer (~16 cm) as a result of mixing caused by cultivation. The ^{137}Cs reference inventories associated with the two identified reference sites (RF1 and RF2) were respectively about 2704 Bq/m^2 and 3554 Bq/m^2. This variation reflects the difference between RF1 and RF2 in mean annual precipitation (about 660 mm and 1700 mm., respectively). ^{137}Cs inventories corresponding to the cultivated sites were the lowest, ranging between 320 and 2998 Bq/m^2, while for the forest and shrub areas, ^{137}Cs activities were found to range between 1340 and 6830 Bq/m^2. For fallout $^{210}Pb_{ex}$, reference inventories associated with the two reference sites were about 3220 Bq/m^2 and 4800 Bq/m^2, respectively for RF1 and RF2. $^{210}Pb_{ex}$ activities were measured for some fields (FR1, SR1, AG1) near to the first reference site (RF1). The mean $^{210}Pb_{ex}$ activity associated with cultivated site AG1 was 2 685 Bq/m^2, while for the forest (FR1) and shrub (SR1) fields the mean inventories were found to be respectively 3 088 and 2950 Bq/m^2.

Medium-term rates (over 50 years) of water-induced soil erosion and deposition were estimated from the ^{137}Cs inventories and using the refined mass balance II model (MBII) for the cultivated site and the diffusion and migration model (DMM) for the uncultivated sites (Walling, He and Appleby, 2002). For the cultivated fields, all the sampling points were found as eroded areas. Consequently, the gross, mean and net soil erosion rates were similar indicating sediment delivery ratios of about 100 percent. Therefore, erosion rates corresponding to agricultural fields ranged between 18 t·ha^{-1}·yr^{-1} and 36 t·ha^{-1}·yr^{-1} (Table 1), while for the forest and shrub areas the net soil erosion rates were found to be low.

Using the DMM model, the net soil erosion rates ranged between 0.8 t·ha^{-1}·yr^{-1} and 2.4 t t·ha^{-1}·yr^{-1} and sediment delivery ratios varied from 52 to 94 percent. In all cases, these results show that soil erosion rates associated with agricultural fields were significantly higher than those associated with the forest and shrub fields. They also demonstrate clearly the role of forest plantations and vegetation cover in protecting the soil against erosion, reducing by a factor of between 4 and 20 the soil loss in the watershed. Until now, net soil erosion rates have been estimated for ten homogeneous units or iso-sectors (Table 1). The extrapolation of data on soil erosion to the watershed scale covering over 72 percent of the total area, indicates an average net soil erosion rate of 22.1 t·ha^{-1}·yr^{-1}. The most vulnerable soils corresponded to the agricultural iso-sector with aridisols and slopes greater than 30 percent (Table 1). These are estimated to have contributed 56 460 t of soil loss annually, representing almost half of the total loss in the area of the watershed investigated. Total annual soil erosion from all agricultural fields was estimated to be

TABLE 1. Net soil erosion rates associated with ten iso-sectors of the watershed

Site	Iso-sector (%)	Area (ha)	% of total area	Net erosion rate (t·ha^{-1}·yr^{-1})	Net erosion rate (t·ha^{-1}·yr^{-1})	% of total erosion
AG1/AG2	AG, VE, 5–15	654	8.8	27.4	17 920	14.0
AG3	AG, AR, 15–30	1 723	23.1	18.5	31 876	24.9
AG4	AG, VE, 15–30	382	5.1	22.5	8 595	6.7
AG5	AG, AR, > 30	1 564	21.0	36.1	56 460	44.1
AG6	AG,AR, 5–15	579	8.4	18.8	10 884	8.5
FR1	FR, AL, > 30	121	1.6	1.6	194	0.2
FR2	FR, AL, 15–30	146	2.0	0.8	117	0.1
SR1	SR, AR, > 30	472	6.3	2.4	1 133	0.9
SR2	SR, AL, 15–30	23	0.3	4.5	104	0.1
SR3	SR, AR, 15–30	119	1.7	6.9	821	0.6
Total		5 783	71.9	22.1	128 121	100

AG — agriculture; FR — forest; SR — shrub; VE — vertisol; AL — alfisol; and AR — aridisol

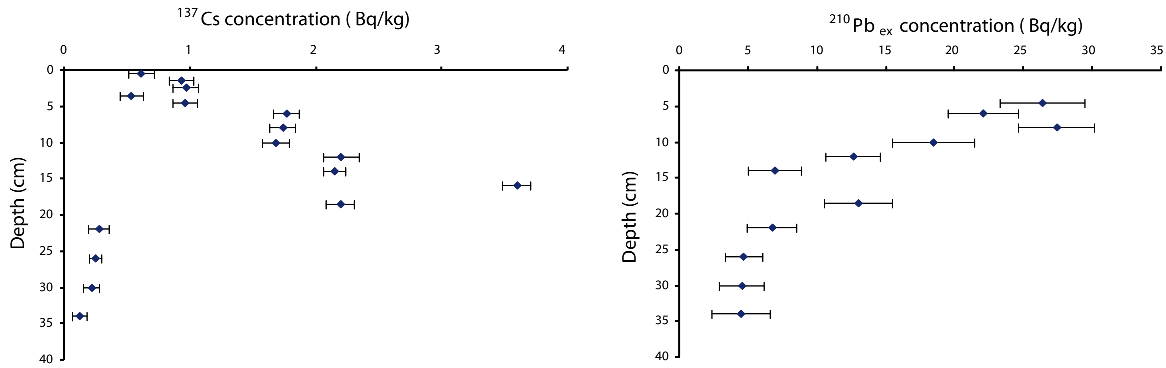

FIGURE 4. ^{137}**Cs and** ^{210}**Pb depth profiles associated with a sediment core collected in the Talembout water reservoir.**

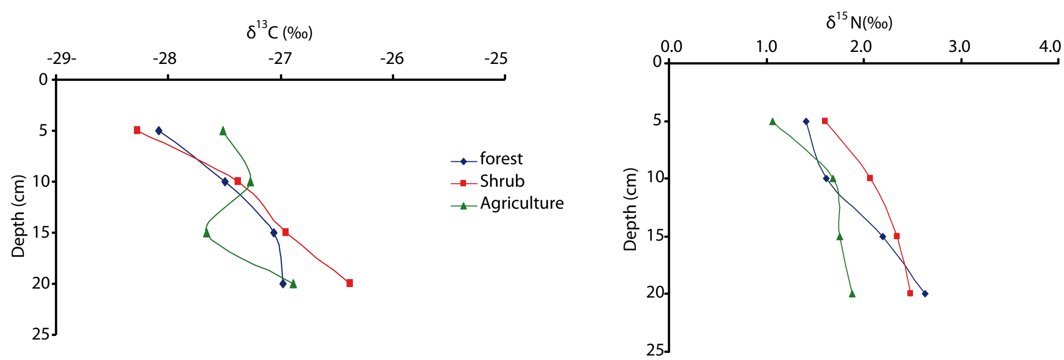

FIGURE 5. Mean depth profiles of δ^{13}**C and** δ^{15}**N associated with forest, shrub and agricultural fields.**

114 818 t. Fallout $^{210}Pb_{ex}$ was also used to estimate long-term soil erosion rates over 100 years in the fields AG1, FR1 and SR1. Preliminary results indicated that the gross erosion rates were respectively about 12 t·ha^{-1}·yr^{-1}, 0.61 t·ha^{-1}·yr^{-1} and 1.3 t·ha^{-1}·yr^{-1}. These values are lower than those obtained by ^{137}Cs, a difference that may be explained by the significant increase in soil loss that has taken place in the watershed during the last 50 years.

The mean sedimentation rate in the water reservoir Talembout is provided in Figure 4. The maximum depth of the ^{137}Cs profile was 16 cm which can be attributed to the atmospheric fallout from nuclear weapon tests occured in 1963. The sedimentation rate was estimated approximately at 0.34 cm/yr. Taking into account the soil bulk density, the sedimentation rate in the water reservoir was about 0.503 g·cm^{-2}·yr^{-1} (50 t·ha^{-1}·yr^{-1}). Using fallout $^{210}Pb_{ex}$ and the constant flux and constant sedimentation (CFCS) model (Krishnaswami *et. al.*, 1971; Appleby and Oldfield, 1978), the sedimentation rate was about 0.495 g·cm^{-2}·yr^{-1} (49.5 t·ha^{-1}·yr^{-1}), i.e. similar to the value obtained by fallout ^{137}Cs. The sedimentation rate was therefore higher than the net erosion rate estimated from the upland fields of My Bouchta watershed. This could be explained by the contributions of other possible forms of water erosion such as gully erosion and mass movement including landeslides.

Stable isotopic signatures (δ^{13}C and δ^{15}N) and C and N contents of soil and sediment samples

The soil δ^{13}C and δ^{15}N signatures as well as carbon and nitrogen concentrations were determined for three different fields on the upland areas of the watershed: forest (AG1), shrub (SR1) and agriculture (AG1). The δ^{13}C associated with forest (AG1), shrub (SR1) and agriculture (AG1) along one transect and at six points of each field ranged from –27.26 to –28.80‰, from –25.71 to –28.60‰ and from –26.51 to –28.34‰, respectively. These values are associated with C_3 plants. Similarly, the δ^{15}N corresponding to the fields ranged from 0.74 to 3.90‰, and from 0.58 to 4.76‰ and from 0.36 to 1.85‰, respectively.

The mean depth profiles of δ^{13}C and δ^{15}N for the three fields are given in Figure 5. The mean value for six points was taken for each level. δ^{13}C and δ^{15}N profiles associated with forest and shrub areas seemed to have the same behaviour and differed from the δ^{13}C and δ^{15}N profiles associated with the agricultural field. Stable isotopic signatures increased with depth especially for the forest and shrub areas. Comparable results were obtained in a study conducted in lower Montane, Ecuador (Rhoades *et al.*, 2000) who recorded a 2–3 percent enrichment of soil δ^{13}C with soil depth. According to these authors, the greater isotopic enrichment at depth was associated with burial of organic matter. The mean depth profiles of C and N showed that the average C content decreased with depth in forest, shrub and agricultural fields, whereas with the N depth profile, mean values deacreased only in the forest and shrub areas, being almost constant for the agricultural field (Figure 6).

Mean values for the stable isotopes δ^{13}C and δ^{15}N as well as for C and N content and C:N ratios including all points for the three fields are given in Table 2. Carbon and nitrogen concentrations and C:N ratios are in general indicative of soil organic matter quality and provide metrics of fertility. Differences were observed especially for

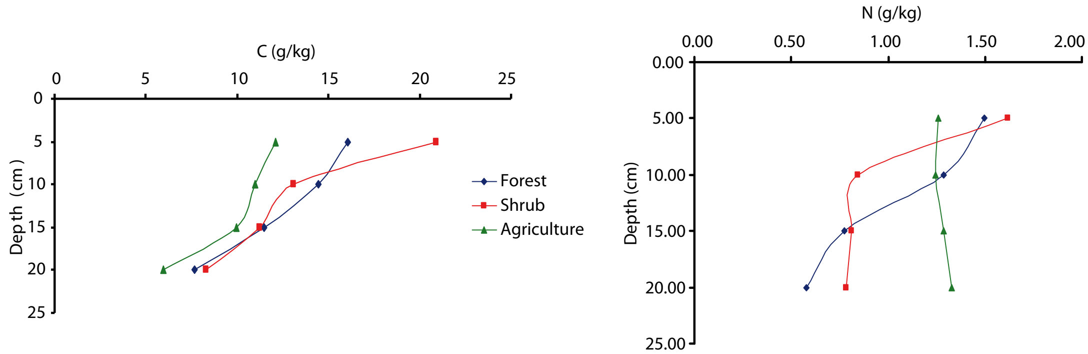

FIGURE 6. Mean depth profiles of C and N contents associated with forest, shrub and agricultural fields.

TABLE 2. Mean values for $\delta^{13}C$, $\delta^{15}N$, C, N and C:N ratios

Field	$\delta^{13}C$ (‰)	C (g/kg)	$\delta^{15}N$ (‰)	N (g/kg)	C:N
Forest (FR1)	–27.49 ± 0.18	12.99 ± 1.39	1.76 ± 0.75	1.04 ± 0.14	12.49± 1.94
Shrub (SR1)	–27.24 ± 0.23	13.05 ± 2.49	2.12 ± 1.00	1.02 ± 0.10	12.79± 2.74
Agriculture (AG1)	–27.33 ± 0.40	9.77 ± 3.96	1.59 ± 0.52	1.28 ± 0.04	7.63 ± 3.10

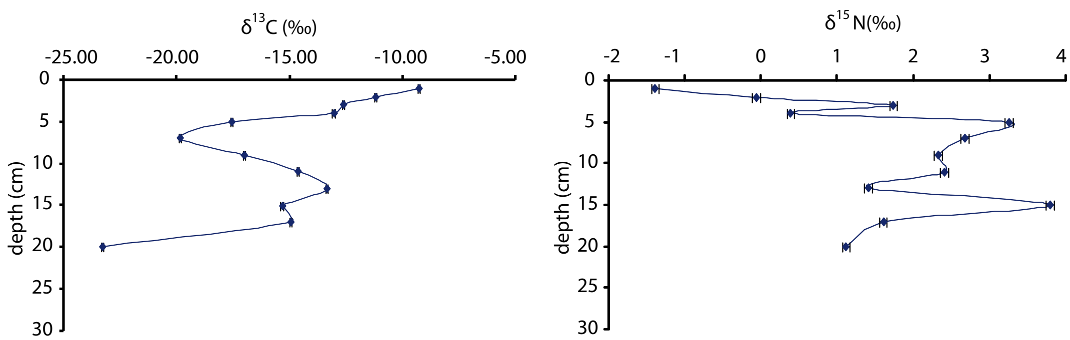

FIGURE 7. $\delta^{13}C$ and $\delta^{15}N$ depth profiles associated with a sediment core collected from the Talembout water reservoir.

the C content and C:N ratios which were higher for forest and shrub soils compared with agricultural soils.

The high correlation between $\delta^{13}C$ and C for forest (R^2 = 0.60) and shrub field (R^2 = 0.74) reflects a linear decrease in C content with an increase in $\delta^{13}C$. However, the correlation for the agricultural field was low (R^2 = 0.24). This can be explained by the high rates of organic matter decomposition arising from high erosion and periodic disturbance of the soil in the agricultural field, compared with the forest and shrub fields. Similar results were obtanied by Schaub and Alwell (2009).

On the other hand, depth profiles of $\delta^{13}C$, $\delta^{15}N$ and the C and N contents associated with one sediment core collected from Talembout water reservoir downstream of My Bouchta watershed (Figures 7 and 8) showed that $\delta^{13}C$ increased from the bottom (–23.25‰ at a depth between 17 and 20 cm) to the top (–9.26‰ at depth of between 0 and 2 cm). The mean value was about –15.15‰. Except for the value obtained at the lower depth which is characteristic of C_3 plants, all other values were higher than –20‰ indicating that the $\delta^{13}C$ signature values were derived from plants within the C_4 group.

These values differed according to the three types of fields (agriculture, forest and shrub) investigated in the uppper areas of the watershed which were derived from C_3 plants. Two interpretations can be provided at this stage:

- The main source of sediment deposits in the water reservoir was soil eroded from the upland areas cultivated by C_4 plants (e.g. maize which is cultivated alternatively with wheat).
- The historic cultivation of maize by farmers around the Talembout water reservoir.

Concerning the depth profile of $\delta^{15}N$, a rapid decline of $\delta^{15}N$ with depth was observed over the last 5 cm (Figure 7). This may be attributed to the increasing use of synthetic inorganic fertilizers on agricultural fields upstream of the reservoir during the last decade. Similar results were obtained by Oczkowski *et al*. (2011) for the sediment cores collected in Lake Manzala (Egypt) and Ghar El Melh Lagoon (Tunisia).

Carbon and N depth profiles are presented in Figure 8. They were maximum at a depth around 7 cm. The variations may reflect changed agricultural practices and fertilizer types.

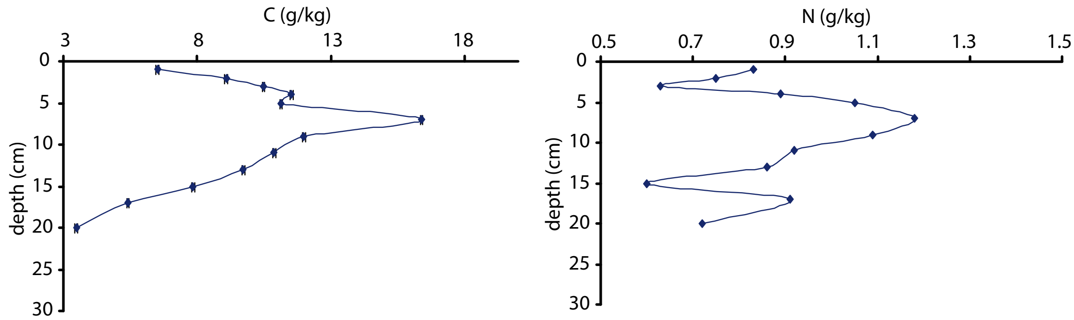

FIGURE 8. C and N depth profiles associated with a sediment core collected from the Talembout water reservoir.

CONCLUSIONS

These preliminary results show that the net soil erosion rate estimated for the My Bouchta watershed was about 22 t·ha^{-1}·yr^{-1}. Rates associated with agricultural fields were significantly higher than those for forest and shrub lands and indicate clearly the role of forest plantations and vegetation cover in protecting soils against erosion. A high sedimantation rate (about 0.50 g cm^{-2} yr^{-1}, (equivalent to 50 t·ha^{-1}·yr^{-1}) was obtained for the Talembout water reservoir where sediment deposits seemed to have originated mainly from fields cultivated with cereals, particularly maize (a C_4 plant).

There is an urgent need to develop new soil conservation strategies and new agricultural practices in the watersheds of northern Morocco to ensure sustainable crop production in agricultural landscapes and to protect water reservoirs against siltation. Results reported here from the use of stable N isotope indicate increasing use of synthetic inorganic fertilizers on upstream agricultural fields during the last decade. This work will be continued by using ^{7}Be as a tracer of short erosion events and compound-specific stable isotopes (CSSI) to identify and apportion with more precision, sediment source areas and erosion hotspots.

AKNOWLEDGMENT

This study was performed as part of a Research Contract within the framework of the FAO/IAEA Coordinated Research Project entitled Integrated Isotopic Approaches for an Area-wide Precision Conservation to Control the Impacts of Agriculture on Land Degradation and Soil Erosion. The authors thank the IAEA for the financial support provided under this contract.

REFERENCES

Appleby, P.G. & Oldfield, F. 1978. The calculation of lead-210 dates assuming a constant rate supply of unsupported ^{210}Pb to the sediment. *Catena,* 5: 1–8.

Belkhiri, A. 1988. *Conséquence de la degradation des basins versants sur les retenues de barrages.* Rapport du séminaire national sur l'aménagement des bassins versants "diagnostic de la situation actuelle", MARA., T.P., PNUD/FAO, 18–28 January, Rabat, 23 pp.

Bouhlassa, S., Moukhchane, M. & Aiachi, A. 2000. Estimates of soil erosion and deposition of cultivated soil of Nakhla watershed, Morocco, using ^{137}Cs technique and calibration models. *Acta Geol. Hisp.,* 35: 239–249.

Gibbs, M.M. 2008. Identifying source soils in contemporary estuarine sediments: A new compound-specific isotope method. *Estuar. Coast.,* 31: 344–359.

IAEA. 2011. Estimates of long and short-term rates of soil erosion using ^{137}Cs, ^{210}Pbex and ^{7}Be measurements: Case study of one agricultural field in semi-arid west Morocco, by M. Benmansour, A. Nouira, H. Bouksirat, M. Duchemin, M. El Oumri, R., Mossadek, A. Benkdad & M. Ibn Majah. In *Impact of soil conservation measures on erosion control and soil quality*, IAEA-TECDOC 1665, pp. 159–174. Vienna.

Krishnaswami, S.D., Lal, D., Martin, J.M. & Meybeck, M. 1971. Geochronology of lake sediments. *Earth Plan. Sci. Lett.,* 11: 407–414.

Mabit, L., Benmansour, M. & Walling, D.E. 2008. Comparatives advantages and limitations of ^{137}Cs, ^{210}Pb and ^{7}Be methodologies for assessing erosion and sedimentation. J. Environ. Radioact., 99: 1799–1809.

Namr, K.L. & Mrabet, R. 2004. Influence of agriculture management on chemical quality of a clay soil of semi-arid Morocco. *J. Afr. Earth Sci.*, 39: 485–489.

Nouira, A., Sayouty, E.H. & Benmansour, M. 2003. Use of ^{137}Cs technique for soil erosion study in the agricultural region of Casablanca in Morocco. *J. Environ. Radioact.,* 68: 11–26.

Oczkowski, A.J., Flower, R.J., Thompson J.R., Ayache F., Ahmed, M.H., Ramdani M. & Turner, S. 2011. Evidence of North Africa's green revolution preserved in sedimentary organic matter deposited in three coastal lagoons. *Ecol. Appl.,* 21: 1708–1717

Phillips, D.L. &Gregg, J.W. 2003. Source partitioning using stable isotopes: Coping with too many sources. *Oecologia,* 136: 261–269.

Rhoades, C., Eckert, G.E. & Coleman D.C. 2000. Soil carbon differences among forest, agriculture and secondary vegetation in lower Montane Ecuador. *Ecol. Appl.,* 10: 497–505.

Schaub, M. & Alwell, C. 2009. Stable carbon isotopes as an indicator for soil degradation in alpine environment (Ur Seren Valley, Switzerland). *Rapid Commun. Mass Spectrom.,* 23: 1499–1507.

Walling, D.E., He, Q. & Quine, T.A. 1995. Use of ^{137}Cs and ^{210}Pb as tracers in soil erosion investigations. *In* C. Leibundgut, ed. *Tracer technology for hydrological systems.* IAHS Publication No. 229, pp. 163–172. Wallingford, Oxford, IAHS Press.

Walling D.E., He, Q. & Appleby, P.G. 2002 Conversion models for use in soil erosion, soil redistribution and sedimentation investigations, *In* F. Zapata, ed. *Handbook for the assessment of soil erosion and sedimentation using environmental radionuclides*, pp. 111-162. Dordrecht, The Netherlands, Kluwer Academic Publishers.

Zapata, F. ed. 2002. *Handbook for the assessment of soil erosion and sedimentation using environmental radionuclides*. Dordrecht, The Netherlands, Kluwer Academic Publishers, 219 pp.

Use of Fallout Caesium-137 to Evaluate the Effectiveness of the FAO-LADA Approach for Assessing Soil Erosion-Induced Land Degradation in the Chinese Loess Plateau

Y. Li [1, 2*], H.Q. Yu[1], Y.Z. Zhang[1], B. Nyamdavaa[1] and G.Q. Liu[1]

ABSTRACT

The Land Degradation Assessment of Drylands methodology (LADA) has been reported to be a rapid and robust approach for assessing the state and nature of soil erosion at regional, national and global levels. However, it needs to be adapted for use under different environments. The studies reported here were to: (i) compare differences in soil redistribution rates determined by the LADA approach and by using the fallout caesium-137 (^{137}Cs) technique, and (ii) develop a validated LADA methodology to rapidly assess soil erosion-induced land degradation for use in the Chinese Loess Plateau. Forty two sites in three watersheds were selected (Liudaogou in Shenmu, Jiuyuangou in Suide, and Nianzhuang in Yan'an) in the Loess Plateau. Soil erosion rates were measured in grassland, shrubland, forestland and farmland by the LADA approach through measurements of plant/tree root exposure, a tree mound and build up against barriers, and rills and by the ^{137}Cs technique. Both approaches revealed similar spatial patterns for soil erosion rates, although the differences between rates determined by the two approaches were greater in grasslands than in forestland and shrubland, and rates derived from the LADA approach were lower in forest and grasslands but higher in farmland than those calculated by the ^{137}Cs technique. It is concluded that the LADA methodology is very useful for quickly assessing soil erosion under different hillslopes and land use types, although an area-wide evaluation of its suitability is still required.

Key words: *caesium-137 technique, LADA, soil erosion, Chinese Loess Plateau*

INTRODUCTION

The Loess Plateau located in west China is perhaps the area in the world most severely affected by soil erosion and deposition. It is estimated that about five billion tonnes of surface soil have been lost due to water erosion from the Loess Plateau, which accounts for 10 percent of total world soil loss. Inappropriate land use and intensive agricultural production with poor farming practices are responsible for this state of affairs. This accelerated soil erosion is a major force driving land and soil quality degradation, and is therefore a threat not only to China, but also to global food security and environmental sustainability (Li, 1995; Li and Lindstrom, 2001). However, the on-site and off-site impacts on productivity and agricultural sustainability of this well-documented acceleration of soil erosion and deposition remain debatable because of the compounding impact of changing climate and the compensating effects of high yields brought about by technological improvements (e.g. new varieties, fertilizer use, soil water conservation). New technologies are therefore urgently needed to develop decision support for sustainable agricultural production and agro-environmental sustainability (IAEA, 2008) based on a clearer understanding not only about the state and nature of soil erosion, but also the impacts of land management at regional, national and global levels.

Changes in soil quality degradation due to soil erosion are major factors to be considered in relation to food security (Lal, 1999). Soil erosion adversely impacts agricultural productivity and the environment through its effects on physical, chemical, and biological quality. Knowledge of these soil quality parameters is key to assessing the impacts of agricultural practices on land degradation in terms of productivity. Over the last 50 yr, soil erosion processes have been well documented in China through runoff plot observations and modelling, but their quantitative impacts on soil quality degradation are largely unknown, especially on an area-wide basis (Li and Lindstrom, 2001; Li *et al.*, 2004). So far no national map exists on soil erosion-induced land degradation, or on soils that have been restored through Sustainable Land Management (SLM) practices adopted over the last decades. In fact, simply by focusing solely on the negative trends is unlikely to provide the detailed understanding of the drivers and impacts of changes in land resources necessary to assess current and improved management and policy responses.

The recently developed Land Degradation Assessment (LADA) methodology by FAO and partners (LADA, 2009a and b), including its parameters for site selection and visual indicators for detailed assessment of soil erosion, is a rapid but robust assessment methodology. It offers the potential for a team of approximately five people with multi-sectoral expertise to implement land degradation assessment over a period of some two to three weeks, including time for analysis and report writing. Previous land degradation assessments have rarely moved much beyond descriptions and quantification of biophysical processes. The LADA methodology, on the other hand, aims to provide an understanding both of the state and nature of changes in land resources (soil, water, vegetation) and ecosystems, and also

1 Institute of Environment and Sustainable Development in Agriculture, Chinese Academy of Agricultural Sciences, Beijing, 100081, P.R. China
2 College of Life Sciences, Hennan Normal University, Henan, Xinxiang 453000, P.R. China
* E-mail address of corresponding author: liyong@caas.cn

L.K. Heng, K. Sakadevan, G. Dercon and M.L. Nguyen (eds), Proceedings — International Symposium on Managing Soils for Food Security and Climate Change Adaptation and Mitigation. Food and Agriculture Organization of the United Nations, Rome, 2014: 119–123

of the drivers and impacts of land degradation LD/SLM. This methodology does not require substantial laboratory-based measurements but provides accuracy and validity by combining quantitative and semi-quantitative field measurements with qualitative information from local stakeholders. Its primary emphasis is on the assessment of the current status and dynamics of land resources — soil, water and vegetation in delivering the main productive services required by land-users from the land, i.e. the provisioning services. A second important consideration concerning the LADA approach is the need to identify and evaluate significant impacts of LD/SLM on other key ecosystem services, particularly on global-level systems and resources such as biodiversity etc. Nevertheless, the LADA methodology needs to be tested and compared with other approaches for determining soil erosion on detailed assessment sites so that it may be adapted for use under different environments.

Steep slopes and dense gullies with intensive crop production and over-grazing activities characterize the Chinese Loess Plateau. This emphasizes the existence of spatial variability of soil erosion-deposition and its impacts on soil quality in terms of productivity and environmental sustainability. Fallout radionuclides (FRNs) including ^{137}Cs, lead-210 (^{210}Pb) and berrylium-7 (^{7}Be) are useful for studying soil redistribution patterns within hillslope landscapes (Wallbrink *et al.*, 1999; Zapata, 2002; Wallbrink, 2004; Walling *et al.*, 2006; Zapata and Li, 2007; Ritchie and Ritchie, 2008) over the short (several months), medium (45-yr span), and long terms (100–150 yr span). Their use should therefore be helpful in developing a validated LADA methodology for assessing soil erosion-induced land degradation rapidly and that can be adapted for use across the main land use and ecosystems in the Chinese Loess Plateau.

The objectives of this study were: (i) to compare differences in soil redistribution rates determined by LADA approaches and ^{137}Cs techniques, and (ii) to develop a validated LADA methodology for assessing soil erosion-induced land degradation rapidly and adapting this for use across the main land use/ecosystems in the Chinese Loess Plateau. Some initial results are presented in this paper.

MATERIALS AND METHODS

Identification of study areas

To facilitate extrapolation of field-level results to the regional level of the entire Loess Plateau, the following three-tiered strategy of site selection was adopted for local assessment using the LADA approach:

(i) Geographic Assessment Area (GAA): Three areas were selected in the Loess Plateau to represent sand loess (Shenmu County), loam loess (Suide City) and loess areas (Yan'an City), respectively (see Figure 1).

(ii) Study Area (SA): One representative watershed was selected within each GAA for the field assessments, i.e. the Liudaogou watershed in Shemu County, the Jiuyuangou watershed in Suide City and the Nianzhuang watershed in Yan'an City.

(iii) Sites/full hillslopes for detailed assessments: 2–3 sites/full hillslopes (SITES) were chosen within each study area representing each land-use type (LUT) — cropland, grassland, forest /woodland — and changes in management practices and biophysical features (land units). The characteristics of each LUT were determined in the SA and are shown in Figure 2.

In total, 108 sites/full hillslopes within each study area (3GAA × 3SA × 3SITES × 4LUTs) were used for detailed assessment of soil erosion and field surveys of which 42 sites have been completed using the LADA approach and FRN techniques (Zapata, 2002).

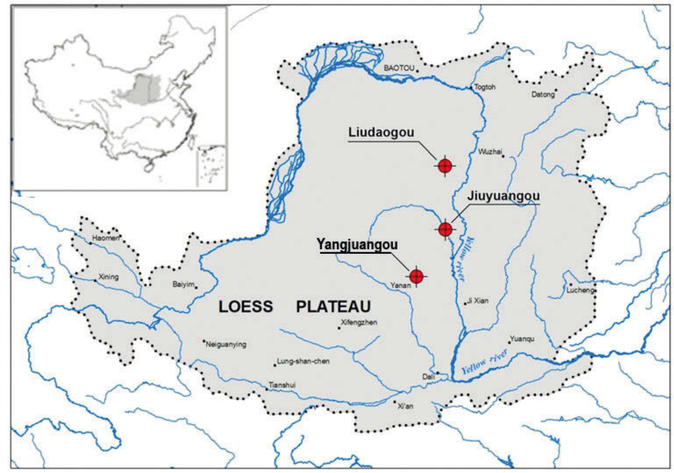

FIGURE 1. Map of the geographic assessment areas.

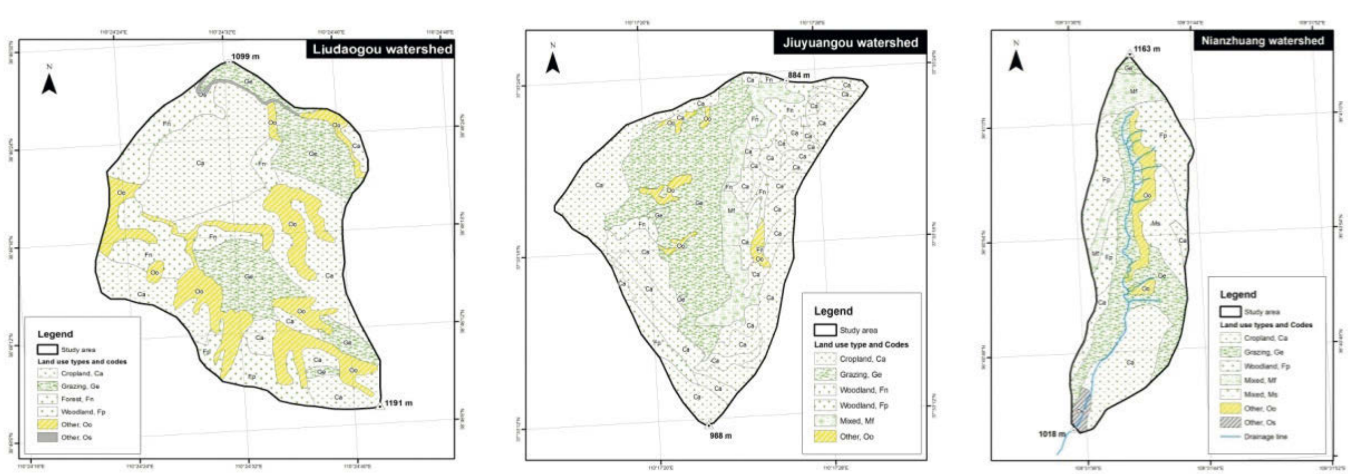

FIGURE 2. Study area (SA): Land use types in the Liudaogou (left), Jiuyuangou (middle) and Nianzhuang watersheds (right).

MEASUREMENT OF EROSION RATES BY LADA APPROACH

Soil erosion rates were measured in grassland, shrubland and forestland including through measurements of plant/tree root exposure, a tree mound and build-up against barriers through procedures and calculations described as follows:

Plant or tree root exposure

The distance from the soil surface to the points on the plant stem or tree trunk which could show the original ground level were measured using a ruler. This measurement gives an estimate of the soil lost since the plant or tree was planted. The annual soil loss ($t \cdot ha^{-1} \cdot yr^{-1}$) was then calculated by dividing the measured difference between the actual soil level and that which existed when the plant or tree started to grow, by the age of the tree. Annual soil loss was calculated using the following formula:

Annual soil loss ($t \cdot ha^{-1} \cdot yr^{-1}$) = Average soil level change (t/ha)/Average age of plants or tree (yr)

Where average soil level change (t/ha) = Average depth of soil loss (mm) × Soil bulk density (t/ha)

Build-up against tree trunk or plant stem

This measurement indicates the accumulation of soil on the upslope side of a tree trunk or plant stem in the target slopes. The volume of soil accumulated was calculated by measuring the depth of the accumulation (at the deepest point) and the distance it extends from the tree trunk or plant stem. For calculation it was assumed that the accumulation was a regular, half-cone shape. The amount of soil accumulated behind a barrier represents the build-up over time, and the eroded material lost in runoff occurring between the trees or plants and this eroded area between the trees or plants along the downslope will have been the contributing area. Detailed calculations were as follows:

(i) Total volume of saved soil

$$\sum \frac{1}{2}\left(\frac{1}{3}\pi r^2 d\right) = \textit{Total volume of soil saved } (m^3)$$

where r (m) — the depth of the accumulation (at the deepest point) and d (m) — the length of accumulation

(ii) The annual rate of soil accumulation (based on the age of trees or plants and land management practices):
Annual volume of soil accumulated (m^3/yr) = Total volume of soil saved (m^3)/Age of trees or plants *(yr)*

(iii) The total volume of soil lost from the estimated contributing area was expressed on the basis of square meters:
Total volume of soil accumulated (m^3/m^2) = Annual volume of soil accumulated (m^3/yr)/Contributing area (m^2), and

(iv) Soil loss rate was calculated using the volume of accumulated soil per square meter and the soil bulk density as below:
Soil lost ($t \cdot ha^{-1} \cdot yr^{-1}$) = Total volume of soil accumulated (m^3 m^{-2}) × Bulk density (t/m^3) ×10 000

To estimate soil erosion rates from the farmland, visible rills induced by concentrated flow erosion events were measured in one year, with the below calculation:

Soil lost ($t \cdot ha^{-1} \cdot yr^{-1}$) = Total volume of rills (m^3) × Soil bulk density (t/m^3) ×10 000/Contributing area (m^2)

Soil erosion rates using ^{137}Cs technique

Soil samples for measuring ^{137}Cs inventories were collected from the same SAs from which the field assessments of soil erosion were carried out using the LADA approach. At the same time, samples were collected for determining reference ^{137}Cs inventories. Details concerning the samples and sites are given in Table 1.

Soil samples were air-dried, weighed, and divided into two parts, one passing through a 0.15-mm sieve for the measurement of soil organic carbon (SOC) concentrations and the other passing through a 2-mm sieve for the measurements of ^{137}Cs activities. Measurements of ^{137}Cs activities were conducted using a hyperpure coaxial Ge detector coupled to a multichannel analyzer (Li *et al.,* 2003). Caesium-137 activity was detected at the 662 keV peak using a counting time of over 80 000 sec, which provided an analytical precision of ± 5 percent (Li *et al.,* 2006).

Using ^{137}Cs to understand soil redistribution involves comparing the measured inventories (total activity in the soil profile per unit area) at all sample sites with an estimate of the total atmospheric input obtained from a "reference site". Also, one can determine whether erosion (less ^{137}Cs present than at the reference site) or deposition (more ^{137}Cs than at the reference site) has occurred. At each sampling point, calculation of soil erosion rates relied on the mass balance model developed by Walling *et al.* (2003).

RESULTS AND DISCUSSION

Soil erosion rates measured using the LADA approach

Table 1 summarizes the land use types and plant species for 42 sites in the SAs while Table 2 shows the results of the soil erosion rates in grassland, shrubland and forestland determined from the surveys carried out on these sites. It is believed that the data in Table 2 represent recent soil erosion rates in the SAs and arose from conversion of grass-shrub- and forestland to farmland since 1998. Noteworthy is that soil erosion rates increased according to the following order under different land uses in the Liudaogou and Jiuyuangou watersheds: grassland (average 0.89 and 0.84 $t \cdot ha^{-1} \cdot yr^{-1}$, respectively) < shrubland (average 5.5 and 4.5 $t \cdot ha^{-1} \cdot yr^{-1}$, respectively) < foreastland (average 6.0 and 6.1 $t \cdot ha^{-1} \cdot yr^{-1}$, respectively). For the Nianzhuang watershed, the rate of erosion was also higher under forestland than grassland. These results can be attributed to the easier identification of soil erosion in forestland than grassland, and the denser root distributions in grassland than in forestland which would enhance significantly the resistance of soil to runoff erosion (Li, 1995). Soil erosion rates on the farmland were estimated by measuring soil lost by rill erosion averaged 74.9 $t \cdot ha^{-1} \cdot yr^{-1}$. This value is 12 to 89 times higher than that measured in the forest and grasslands.

Soil erosion rates estimated using ^{137}Cs inventories

The rate of soil erosion derived from from ^{137}Cs averaged –7.0 $\cdot ha^{-1} \cdot yr^{-1}$ in forest hillslopes over the past 50 years (Figure 1). For the farmland, 18 soil samplings were conducted in the inter-rill and rill areas. From these, the soil erosion rate averaged 42.1 $t \cdot ha^{-1} \cdot yr^{-1}$ using the ^{137}Cs conversion model, much higher than forest and grasslands. This infers that reforestation on the cultivated hillslopes has significantly reduced soil erosion.

Comparing the LADA approach with FRN techniques

The spatial patterns of soil erosion were similar for the LADA approach and ^{137}Cs technique (Figure 1). The somewhat higher value

TABLE 1. Summarized land use types (LUTs) and plant species in the Chinese Loess Plateau

SA	LUTs	Plant species	Conversion of vegetation from farmland (yr)	Number of investigated sites
Liudaogou watershed in Shenmu county	Grassland1	Pennisetum flaccidum Griseb	10	3
	Grassland2	Tripolium vulgare Ness	5	3
	Forestland1	Populussimonii Carr	10	3
	Forestland2	Pinus tabulaeformis Carr	10	3
	Forestland3	Ulmus pumilaL.	10	3
	Shrubland1	Caragana intermedia	5	3
	Shrubland1	Caragana intermedia	10	3
Jiuyuangou watershed in Suide city	Grassland	*Medicago sativa Linn*		3
	Forestland	*Pinus tabulaeformis Carr*	7	3
	Shrubland	*Caragana intermedia*	7	3
	Farmland	Crop (Maize and Potato)	7	3
Nianzhuang watershed, Ya'an city	Grassland	*Pennisetum flaccidum Griseb*		3
	Forestland	*Robinia pseudoacaciaL.*	7	3
	Farmland	Crop (Maize and Potato)	7	3
Total				42

TABLE 2. Soil erosion rates with the LADA approach in the Chinese Loess Plateau

SA	LUTs	Soil erosion (cm)	Eroded period (yr)	Soil bulk density (g/cm)	Soil erosion rate $t{\cdot}ha^{-1}{\cdot}yr^{-1}$
Liudaogou watershed in Shenmu county	Grassland1	5	10	1.46	0.73
	Grassland2	3.8	5	1.37	1.04
	Forestland1	46	10	1.48	6.79
	Forestland2	40	10	1.31	5.24
	Forestland3	35	10	1.68	5.86
	Shrubland1	28	5	1.38	7.72
	Shrubland1	24	10	1.34	3.21
Jiuyuangou watershed in Suide city	Grassland	4.5	7	1.31	0.84
	Forestland	36	7	1.18	6.07
	Shrubland	25	7	1.27	4.52
Nianzhuang watershed in Ya'an city	Grassland	5	7	1.17	0.84
	Forestland	45	7	1.13	7.29

from the LADA approach is indicative of short-term soil erosion rates over 5–10 yr, while the ^{137}Cs technique estimates long-term average soil erosion over the past 50 yr. Overall, however, the results suggest that the LADA approach is very useful for quickly assessing soil erosion on different hillslope positions and land use types.

To compare the differences between the methods, soil redistribution rates determined by ^{137}Cs technique were subtracted from those obtained by the LADA methodology in the three SAs (Figure 2). For the different land use types, the differences between soil erosion rates determined by the LADA approach from those determined by ^{137}Cs were larger for grasslands (average 6.1 $t{\cdot}ha^{-1}{\cdot}yr^{-1}$, respectively) than for forestland (average 6.1 $t{\cdot}ha^{-1}{\cdot}yr^{-1}$) and shrubland (average 1.6 $t{\cdot}ha^{-1}{\cdot}yr^{-1}$). The LADA approach cannot measure soil lost by sheet erosion, and this may explain the larger difference in grassland than in forestland and shrubland. However, in farmland, the rill erosion rate was significantly higher (average 74.9 $t{\cdot}ha^{-1}{\cdot}yr^{-1}$) when measured by the LADA approach compared with using ^{137}Cs (average 42.1 $t{\cdot}ha^{-1}{\cdot}yr^{-1}$). Since little or no ^{137}Cs could be detected, it may be inferred that the ^{137}Cs method underestimated rill erosion.

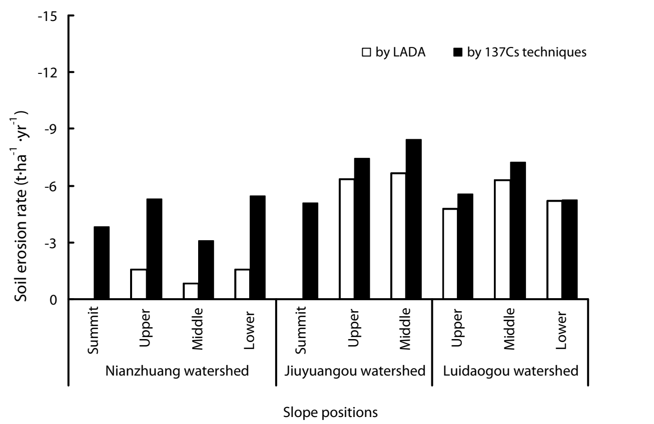

FIGURE 3. Comparison between spatial patterns of soil erosion rates by the LADA methodology and the ^{137}Cs technique in forest hillslopes.

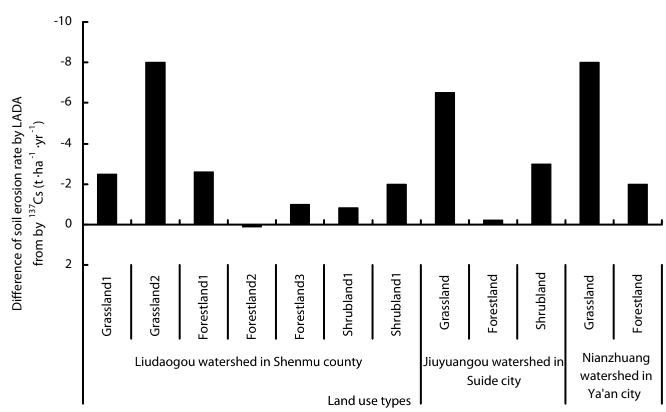

FIGURE 4. Difference in soil erosion rates determined by LADA and FRN techniques.

CONCLUSIONS

The LADA approach is very useful for quickly assessing soil erosion on different hillslope positions and land use types, providing a spatial pattern similar to that obtained using the ^{137}Cs technique; soil erosion rates derived from the LADA approach are, nevertheless, lower than those calculated by the ^{137}Cs technique on forested hillslopes, but higher on cultivated slopes in the Chinese Loess Plateau. Differences found between the methods in erosion rates may be explained by the LADA approach providing an estimation of visible soil loss, while figures obtained with the ^{137}Cs technique include both sheet erosion and inter-rill erosion. Also, the LADA approach yields information about short-term soil erosion rates (5–10 yr), while ^{137}Cs technique provides an estimate of long-term average soil erosion over the past 50 yr.

ACKNOWLEDGEMENTS

This study was supported by the National Natural Science Foundation of China (grant nos. No.41171231, and No.31000944) and by FAO/IAEA Coordinated Research Project (nos. 15479, 15938, 16290 and 16941).

REFERENCES

IAEA. 2008. Report of an FAO/IAEA Consultants Meeting on integrated approaches for the assessment of land use impacts on soil loss/sediment production and related environmental problems, 5–7 November 2007, Vienna. (http://www-naweb.iaea.org/nafa/swmn/public/snl-30-2.pdf).

LADA. 2009a. *Field manual for local level land degradation assessment in drylands, LADA-L Part 1: Methodological approach, planning and analysis,* 76 pp.. (http://knowledgebase.terrafrica.org/fileadmin/user_upload/terrafrica/docs/Local_manual_1.pdf)

LADA. 2009b. *Field manual for local level land degradation assessment in drylands, LADA-L Part 2: local assessment: tools and methods for fieldwork*, 133pp. (http://www.arc.agric.za/uploads/Documents/6653_Local_manual_2.pdf).

Lal, R. ed. 1999. *Soil quality and soil erosion*. . Boca Raton, London, New York & Washington, DC. CRC Press.

Li, Y. 1995. *Plant roots and soil anti-scouribility in Chinese Loess Plateau*. Science Press, Beijing.

Li, Y. & Lindstrom, M.J. 2001. Evaluating soil quality-soil redistribution relationship on terraces and steep hillslope. *Soil Sci. Soc. Am. J.*, 65: 1500–1508.

Li, Y., PoesenJ., Yang, J.C., Fu, B. & Zhang, J.H 2003, Evaluating gully erosion using ^{137}Cs and $^{210}Pb/^{137}Cs$ ratio in a reservoir catchment, *Soil Till. Res.*, 69: 107–115.

Li, Y., Tian, G., Lindstrom, M.J. & Bork, H.R 2004. Variation of surface soil quality parameters by intensive donkey-drawn tillage on steep slope. *Soil Sci.Soc. Am. J.*, 68: 907–913.

Li, Y., Zhang, Q.W., Reicosky, D.C., Bai, L.Y., Lindstrom, M. & Li, L. 2006. Using ^{137}Cs and $^{210}Pb_{ex}$ for quantifying soil organic carbon redistribution affected by intensive tillage on steep slopes. *Soil Till. Res.*, 86: 176–184.

Ritchie, J.C. & Ritchie, C. 2008. *Bibliography of publications of ^{137}Cs studies related to erosion and sediment deposition.* Last updated: October 2007. USDA-ARS.

Wallbrink, P.J. 2004 Quantifying the erosion processes and land uses which dominate fine sediment supply to Moreton Bay, South Queensland, Australia. *J. Environ. Radioact.*, 76: 67–80.

Wallbrink, P.J., Murray, A.S. & Olley, J.M. 1999. Relating suspended sediment to its original soil depth. *Soil Sci. Soc. Am. J.*, 63: 369–378.

Walling, D.E., Collins, A.L. & Sichingabula, H.M. 2003, Using unsupported lead-210 measurements to investigate soil erosion and sediment delivery in a small Zambian catchment, *Geomorphology*, 52: 193– 213.

Walling, D.E., Collins, A.L. Jones, P.A. Leeks G.J.L. & Old, G. 2006. Establishing fine-grained sediment budgets for the Pang and Lambourn LOCAR study catchments. *J. Hydrol*., 330: 126–141.

Zapata, F. ed. 2002. *Handbook for the assessment of soil erosion and sedimentation using environmental radionuclides*. Dordrecht, The Netherlands, Kluwer Publications. 221 pp.

Zapata, F. and Li, Y. 2007. Report of the fourth and final research coordination meeting of the FAO/IAEA coordinated research project "Assess the effectiveness of soil conservation techniques for sustainable watershed management using fallout radionuclides". IAEA-311-D1-RC-888, November 2007, Vienna, Austria.

Quantifying Agricultural Land Degradation Processes Related to Soil Carbon and Nitrogen Redistribution in Western China Using Fallout Radionuclide (FRN) and $\delta^{13}C$ Techniques

Y. Li[1,2,*], H.Q. Yu[1], Y.Z. Zhang[1], B. Nyamdavva[1] and Z.D. Zhang[3]

ABSTRACT

Over the last 50 years, soil erosion rates at field scale have been well documented in China, but knowledge of their quantitative impacts on soil quality is very limited, especially on an area-wide basis. The objective of this study was to quantify the agricultural land degradation processes related to soil carbon (C) and nitrogen (N) redistribution in cultivated slope land of the Chinese Loess Plateau. A cultivated sloping land (a slope gradient of 3.1°–16.8°) in Pucheng County on the Loess Plateau was selected and the history of upland degradation processes reflected by changes in water and tillage erosion and crop productivity was reconstructed through the dated chronology of soil accumulation at deposited sites using fallout radionuclides (FRNs) caesium-137 and excess lead-210 (^{137}Cs and $^{210}Pb_{ex}$), and sources of soil organic carbon (SOC) were identified using $\delta^{13}C$ tracer. The results showed that between 1954 and 2008, bulk soil translocation processes arising from intensive tillage activities were mainly responsible for land degradation in the cultivated slope catchment. Further, changes in crop productivity could be determined from records of C input by crop roots derived from the deposited profile. Models were established using FRN integrated with terrain attributes for slope-catchment evaluation of SOC and total nitrogen (TN) stocks covering a timeframe of 50–100 years. These models had a very high accuracy for quantifying changes in SOC and TN stocks in cultivated slope catchments. Using FRN profile dating in combination with natural $\delta^{13}C$ tracer, it was possible to explain the role of water erosion and intensive tillage processes in affecting upland degradation over the past 100 years.

Key words: *soil erosion, soil quality, fallout radionuclides, profile dating, organic carbon and nitrogen stocks.*

INTRODUCTION

Change in soil quality due to soil erosion is a major factor to be considered in relation to food security (Latta and Lal, 1999). Soil erosion adversely impacts agricultural productivity and the environment through its effects on physical, chemical, and biological quality (Li, Poesen and Valentin, 2004a; Nearing, Pruski and O'Neal, 2004; Lal, 2007). These soil quality parameters are key to assessing the impacts of agricultural practices on land degradation in terms of productivity (Li, 2004b; Wallbrink, 2004**)**. Over the last 50 years, soil erosion rates at field scale have been well documented in China, but understanding of their quantitative impacts on soil quality is very limited, especially on an area-wide basis (Li *et al.*, 2001 and 2004b). So far, no local or national map is available on soil quality degradation due to erosion. Such information is important to restore the quality of eroded soils because of the finite and easily degraded soil resources in China (Li and Lindstrom, 2001). There is increasing evidence that agricultural soil erosion perturbs the global carbon (C) cycle, but the changes in C dynamics induced by accelerated soil erosion and deposition in agricultural landscapes are poorly understood, especially in the steep cultivated hillslope landscapes of west China (Wallbrink, 2004; Li, *et al.,* 2007). The combined use of fallout radionuclide (FRNs) and stable isotopic techniques has the potential for developing quantitative relationships between soil redistribution and soil quality among soils and agricultural ecosystems so that cause-effect relationships can be established for introducing precision conservation measures.

Previous studies suggested that topography plays an important role in agricultural fields in shaping the spatial variability of soil quality parameters through soil redistribution processes (Li and Lindstrom, 2001). Water erosion was the primary cause of the overall decline in soil quality on a steep cultivated hillslope, while tillage erosion had a similar contribution to the overall level of soil quality on a terraced hillslope (Poesen, *et al.,* 1996; Li *et al*., 2006). Soil movement by tillage controlled the spatial patterns of organic matter (OM), N and phosphorus (P) on both terraced and steep cultivated hillslopes (Pennock, 2005). Selective removal of finer particles by water erosion caused a linear decrease in clay content of 0.02%/m, and a corresponding increase in silt content of 0.04%/m downslope of the steep cultivated hillslope (Li and Lindstrom, 2001). However, the relationship between soil redistribution and soil quality parameters needs to be up-scaled from a field scale to catchment and regional scales (IAEA, 2008 unpublished).

The objectives of the study reported here were to use fallout ^{137}Cs and $^{210}Pbex$ in combination with $\delta^{13}C$ to quantify land degradation processes: firstly, by understanding the magnitude and mechanisms of soil C and nutrient changes, and secondly, by reconstructing the evolution of soil organic carbon (SOC) sources in a cultivated slope.

1 Institute of Environment and Sustainable Development in Agriculture, Chinese Academy of Agricultural Sciences, Beijing, 100081, P. R. China
2 College of Life Sciences, Hennan Normal University, Henan, Xinxiang 453000, P. R. China
3 Beijing Normal University, Beijing, 100875, P. R. China
* E-mail address of corresponding author: liyong@caas.cn

L.K. Heng, K. Sakadevan, G. Dercon and M.L. Nguyen (eds), Proceedings — International Symposium on Managing Soils for Food Security and Climate Change Adaptation and Mitigation. Food and Agriculture Organization of the United Nations, Rome, 2014: 125–131

MATERIALS AND METHODS

Study site

The study site was in Pucheng county (35°03′18.45″N, 109°38′25.34″E) of Shaanxi Province, which is in a clay loess area of the Chinese Loess Plateau. A typical sloping land with a slope gradient of 3.1°–16.8° and slope length 54 m at the site was selected for spatial analysis of changes in SOC and TN stocks induced by tillage and water erosion. The site has a westerly monsoon climate; long-term mean temperature is 13.2°C with a range of −16.7 to 42.8°C and precipitation is 540 mm, with 70 percent of annual rainfall occurring between July and September. Soil formed from loess parent materials has a uniform texture (25 percent clay, 70 percent silt, and 5 percent sand). Water and tillage erosion accelerated by poor soil management are responsible for the degradation of soil physical and chemical properties in cultivated slopes at the site. Soil organic C content in the loess soil is between 0.1 and 2.4 percent, but less than 1 percent in most cases. Wheat is the major crop in rotation with corn and sweet potato.

Soil sampling

The spatial patterns of SOC and TN stocks and of ^{137}Cs and ^{210}Pb$_{ex}$ inventories were documented from soil cores taken on a grid of 5 m x 5 m from the cultivated slope catchment using an 8.0 cm diameter hand-operated core sampler. Reference sites for determining ^{137}Cs and ^{210}Pbex inventories were established on a level and undisturbed graveyard with a 100–yr history at the hilltop, less than 30 m from the study sloping land. Soil cores were taken at depths of 0–45 cm at upper and middle positions and at 0–60 cm at the lower positions to ensure that the core had penetrated to the full depth of the ^{137}Cs and ^{210}Pb$_{ex}$ profile. Soil cores were also analysed for bulk densities (g/cm), calculated from the volume of soil cores and oven-dried soil mass. In total, 54 core profiles were collected.

A topographic survey was conducted using a 1-m grid and the coring locations from the slope catchment using a GPS-RTK (Ashtech Z-Xtreme, USA, resolution: 3 mm ± 1 ppm) for development of a digital elevation model (DEM).

Laboratory analyses

Soil samples were air-dried, weighed and divided into two parts, one passing through a 0.15 mm sieve for the measurement of SOC and TN concentrations and the other passing through a 2-mm sieve for measuring ^{137}Cs and ^{210}Pb$_{ex}$ activities. Soil samples for measuring ^{210}Pb$_{ex}$ were sealed in containers and stored for 28 d to ensure equilibrium between ^{226}Ra and its daughter ^{222}Rn (half-life 3.8 d). The amounts of ^{210}Pb$_{ex}$ in the samples were calculated by subtracting ^{226}Ra-supported ^{210}Pb$_{ex}$ concentrations from the total ^{210}Pb$_{ex}$ concentrations. Measurements of ^{137}Cs and ^{210}Pb$_{ex}$ activities were conducted using a hyper-pure coaxial Ge detector (BE5030, Canberra, USA) coupled to a multichannel analyzer. ^{137}Cs activity was detected at 662 keV while total ^{210}Pb concentration was determined at 46.5 keV, and the ^{226}Ra-supported ^{210}Pb was obtained at 609.3 keV using a counting time of over 80 000 seconds. This provided an analytical precision of ±5 percent for ^{137}Cs and ±8 percent for ^{210}Pb. Soil organic C was determined by dry combustion and using an Auto TOC Analyzer (Germany), and measurement of TN used the Kjeldahl digestion method.

Calculations of soil redistribution rates and soil C and TN inventories

At each sampling point, total soil redistribution (TSR) rates due to the combined action of tillage and water erosion were determined for the periods of 1954–2005 and 1857–2007 using ^{137}Cs and ^{210}Pb$_{ex}$ inventories following the mass balance model developed by Walling, He and Appleby (1997) and Walling, Collins and Sichingabule (2003).

Soil organic C and TN stocks over the cultivated slope were calculated by multiplying the concentration of SOC and TN (g/kg) by

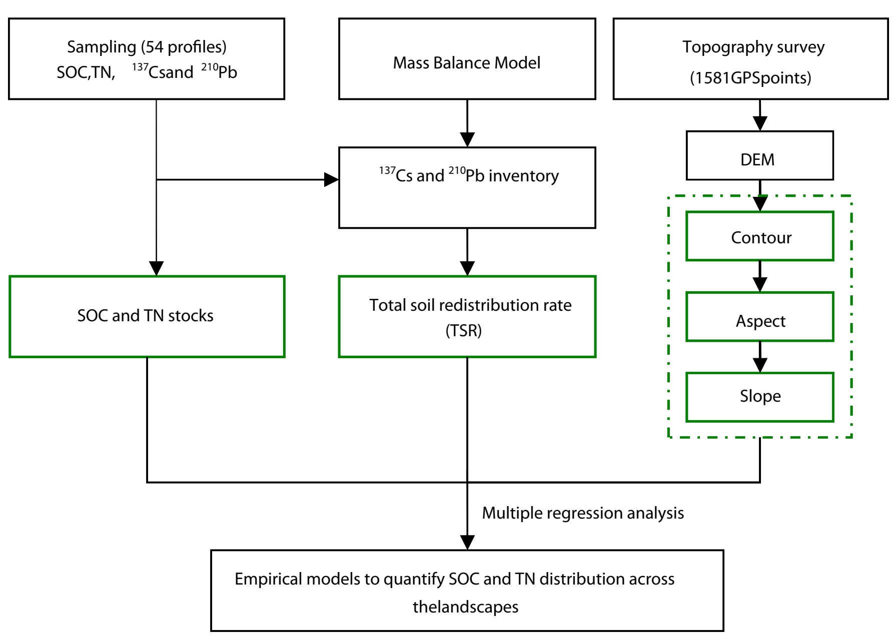

FIGURE 1. Procedure for developing empirical models to quantify SOC and TN distribution across landscapes using multiple regression analysis.

soil bulk density (g/cm) and soil depth (cm) of each sampled soil and expressed in mass per unit area (t·C/ha).

Development of empirical models to quantify soil C and N distribution across the landscapes

The procedure for model development is described in Figure 1. The input parameters include total soil redistribution rates ($t \cdot ha^{-1} \cdot yr^{-1}$) and selected terrain attributes such as relative elevation (E, m), slope gradients (S, in degrees) and slope aspect (A, in degrees). Outputs of the models include SOC and TN stocks (t/ha).

Reconstructing the evolution of land degradation by water and tillage erosion in cultivated slopes by identifying SOC sources at deposited sites

Three methods were adopted for determining the chronology of soil accumulation at deposited sites: using ^{137}Cs and $^{210}Pb_{ex}$ dating; identifying the sources of SOC in the profile of deposited sites using natural $\delta^{13}C$ tracer; and explaining the major factors driving land degradation on the cultivated slope through combined analysis of the first two. These approaches are described below.

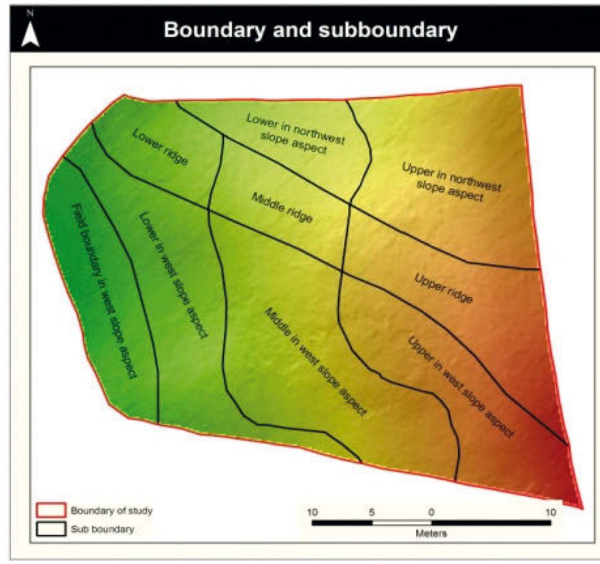

FIGURE 2. Sketch showing boundary and sub-boundary of the study slope catchment derived from the slope aspect.

RESULTS AND DISCUSSION

Spatial patterns of SOC, N, C:N ratios, ^{137}Cs and $^{210}Pb_{ex}$ in slope catchments

In order to analyse the spatial patterns in downslope direction, we divided the slope into three slope locations according to slope gradients: slope gradients ranged 3.1–13.7° for lower slope (slope length 13 m), 8.7–16.8° for middle slope (slope length 21 m), and 4.9–14.0° for upper slope (slope length 20 m). Soil organic C, total N (TN) and C:N ratios displayed similar spatial patterns to ^{137}Cs and $^{210}Pb_{ex}$ inventories in the slope catchment. In the ridge of the slope catchment (Figure 2), both ^{137}Cs and $^{210}Pb_{ex}$ inventories increased in the downslope and changed according to the slope location: lower slope > middle slope > upper slope. Reference values calculated for ^{137}Cs and $^{210}Pb_{ex}$ inventories were 1267 ± 132 Bq/m^2 and 10 675 ± 2599 Bq/m^2, respectively. Compared with the reference site, ^{137}Cs inventory losses were respectively 91.9 percent, 88.8 per cent and 70 percent at the upper, middle and lower slopes over the last 50 years. By contrast, compared with the reference site, $^{210}Pb_{ex}$ losses (which accounted for 50 percent of the total losses of ^{137}Cs inventories), were respectively 48.7 percent, 44.0 percent, and 7.7 percent at the upper, middle and lower slopes over the last 100 years.

Similarly, in the west slope aspect both ^{137}Cs and $^{210}Pb_{ex}$ inventories increased in the direction of the downslope and according to the following slope location: field boundary > lower slope > middle slope > upper slope. Compared with the reference site, ^{137}Cs inventory losses were respectively 89.9 percent, 75.4 percent and 54.8 percent at the upper, middle and lower slopes, while ^{137}Cs inventory gain was 17.6 percent at the field boundary. For $^{210}Pb_{ex}$, losses were respectively 43.8 percent, 38.7 percent, 23.1 percent and 5.2 percent at the upper, middle and lower slopes and at the field boundary. At the northwest slope aspect, ^{137}Cs and $^{210}Pb_{ex}$ inventory losses were respectively 83.2 percent and 54.5 percent.

Like ^{137}Cs and $^{210}Pb_{ex}$, SOC, TN and C:N ratios decreased in the order of slope location in both ridge direction and west slope aspects, but the highest values for SOC and TN were found at the lower field boundary of the slope catchment whereas much higher C:N ratios were found at the lower slope location than at the lower field boundary and other slope locations. For the northwest slope aspect, SOC, TN and C:N ratios were much higher at the lower slope than at the upper slope location. These spatial patterns of SOC, TN and C:N distribution suggest the existence of a mechanism associated with topographical attributes that controls soil redistribution arising from soil use and management practices.

Impacts of topography on distribution of SOC, N and C:N ratios

The influence of topography on SOC and TN was assessed by simple linear regression analysis of the relationship between SOC and TN stocks and topographical attributes (relative elevation, slope degree and slope aspect) (Table 1). The SOC and TN stocks were correlated negatively with both relative elevation ($p < 0.001$) and with slope ($p < 0.01$ and $p < 0.05$, respectively). However, while SOC stocks did not correlate significantly with slope degree, TN stocks did so ($p < 0.05$). These findings indicate that SOC and TN distribution were controlled by the relative elevation and slope aspect of the catchment.

Table 1. Relationships between slope relative elevation (E, m), slope degree (S), slope aspect (A) and SOC (t/ha) and TN (t/ha)

Linear regression	R^2	n	p
$SOC_{stock} = -1.7067E + 29.415$	0.6606	54	<0.001
$SOC_{stock} = -0.0685A + 42.018$	0.1354	54	<0.01
$SOC_{stock} = -0.2627S + 25.596$	0.0278	54	n.s.*
$TN_{stock} = -0.1088E + 2.7229$	0.4119	54	<0.001
$TN_{stock} = -0.0048A + 3.6425$	0.1012	54	<0.05
$TN_{stock} = -0.041S + 2.7364$	0.1036	54	<0.05

*Not significant

Slope catchment models to quantify SOC and TN stocks in cultivated landscapes

Table 2 shows the empirical models established for slope-catchment prediction of SOC and TN stocks in relation to soil redistribution and terrain attributes following the procedures listed in Figure 1.

Comparing the values measured with those estimated by the models from ^{137}Cs and $^{210}Pb_{ex}$, the relative errors for SOC stocks were less than 10 percent for 28 of the 54 observation points whereas those for TN were less than 10 per cent for 27 of 54 observations and 35 of 54 observations, respectively. For the 54 observation points, the average relative errors for SOC and TN were 9.5 percent and 8.3 percent respectively. This suggests that these models have a very high accuracy for quantifying changes in SOC and TN stocks in cultivated slope catchments.

Because the regression coefficients of terrain attributes in the models are relatively stable compared with soil redistribution rates, the models in Table 3 could be used to assess the effectiveness of soil use and management practices on SOC and TN over a time scale of 50–100 yr. For example, over the period 1954–2007, net SOC and TN losses from the slope due to soil erosion were calculated to be 0.36 t·ha^{-1}·yr^{-1} and 0.14 t·ha^{-1}·yr^{-1}, respectively, indicating significant decreases in SOC and TN storage. In contrast, for the period 1907–1954, net SOC and TN gains due to soil erosion were respectively 0.204 and 0.027 t·ha^{-1}·yr^{-1}, indicative of noticeable increases in SOC and TN storage during this period.

However, much work remains to be done for an area-wide evaluation of agricultural land degradation induced by soil erosion. Future focus will therefore be directed at upscaling these slope catchment models from a sandy loess area to silt and loessial areas of the Chinese Loess Plateau.

Evolution of land degradation reconstructed by identifying SOC sources at deposited sites by application of FRNs and $\delta^{13}C$

Chronology of soil accumulation at deposited sites using ^{137}Cs and $^{210}Pb_{ex}$ dating

Two typical profiles were selected at both eroded and deposited sites of the cultivated slope catchment and ^{137}Cs and $^{210}Pb_{ex}$ were measured in both profiles (Figure 3). ^{137}Cs and $^{210}Pb_{ex}$ dating techniques were used to determine the chronology of soil accumulation

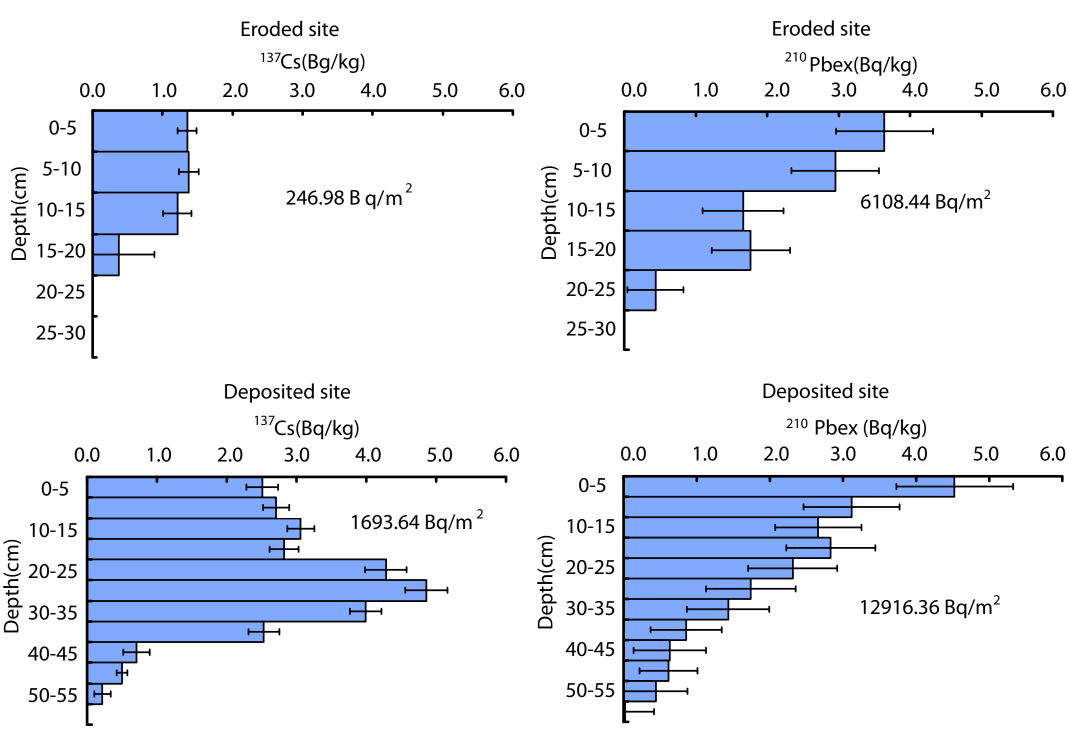

FIGURE 3. ^{137}Cs and $^{210}Pb_{ex}$ profiles at eroded and deposited sites of a cultivated slope catchment.

Table 2. Empirical models for predicting SOC and TN stocks at slope catchment scale using relative elevation (E), slope gradients (S), slope aspect (A) and total soil erosion (TSR)

Period	Multiple regression model	Correlation coefficient (R)	p value	Average % deviation from observed value
1954–2007	SOC_{stock}=44.34–1.49E–0.04A–0.38S+0.006TSR	0.8613	< 0.01	9.4
1907–2007	SOC_{stock}=45.14–1.53E–0.04A–0.39S+0.016TSR	0.8603	< 0.01	9.6
1954–2007	TN_{stock}=3.81–0.065E–0.002A–0.041S+0.002TSR	0.8135	< 0.01	8.6
1907–2007	TN_{stock}=3.75–0.071E–0.002A–0.036S+0.012TSR	0.8351	< 0.01	8.0

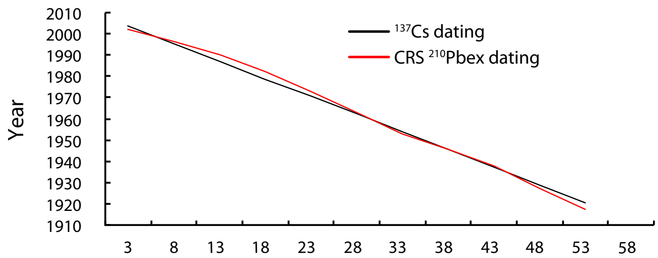

FIGURE 4. Radiometric chronology of soil accumulation profile at deposited site showing agreement between $^{210}Pb_{ex}$ CRS dates and ^{137}Cs dates.

at deposited sites of the slopes, while the $\delta^{13}C$ values of bulk soil and SOC were determined to identify the source of SOC in the profiles.

In the profile from the upland eroded site (Figure 3), ^{137}Cs was distributed uniformly within 0–15 cm of the plough layer whereas $^{210}Pb_{ex}$ decreased with soil depth, although this was also mainly distributed within 0–20 cm. By contrast, a peak of ^{137}Cs activity was found at depths between 25 and 30 cm (Figure 3) in the profile at the deposited site. This can be equated with the maximum atmospheric fallout in 1963. $^{210}Pb_{ex}$ activity decreased with soil depth and was still detected at a depth of 55 cm, i.e. much deeper than that at the eroded site. The temporal patterns of soil accumulation since 1927 at the deposited site of the cultivated slope were reconstructed using ^{137}Cs peak serigraphy in combination with CRS (constant rate of ^{210}Pb supply) $^{210}Pb_{ex}$ Dating Model (Figure 4). The identical chronology determined by these two independent methods suggests that it is possible to date recent soil accumulation with a high degree of accuracy, especially during the past 50 yr.

Profile distribution of SOC sources at deposited site of cultivated slope catchment using natural $\delta^{13}C$ tracer

Based on the data from Figure 5, SOC sources at deposited sites—were calculated using the following formulae:

$$Ce + Cr = Ct \quad (1)$$

$$\delta^{13}Ce \times Ce + \delta^{13}Cr \times Cr = \delta^{13}Ct \times Ct \quad (2)$$

where Ce (g/kg) — eroded soil C from upland; Cr (g/kg) — C input by crop roots; and Ct (g/kg) — total C in the deposited soil profile.

Higher values for Ce than for Cr were observed between 0 and 30 cm., indicating a major contribution to SOC sources by eroded SOC from upland (Figure 6). By contrast, at 30–60 cm depth, values for Ce were lower, indicating that crop inputs provided the major sources of SOC.

Dominant soil redistribution processes controlling upland degradation of slope catchment

To understand more fully the upland degradation processes induced by water erosion and intensive tillage activities, changes in the SOC content from eroded SOC and in inputs of crop roots were estimated by linking the chronology of soil accumulation (Figure 4) and SOC sources in deposited profiles at the lower field boundary of the slope (Figure 6).

It is clear that values for Ce and Cr showed distinct changes between the years 1927 and 1954 and between 1954 and 2008 (Figure 7). Values for Ce during the period 1927 to 1954 were much lower than between 1954 and 2008, ranging from 0.56 to 3.36 g/kg with a mean of 2.03 g/kg compared with between 2.56 and 8.47 g/kg with a mean of 5.85 g/kg during the period 1954 and 2008, i.e. around a two-fold increase during the latter period.

Recent studies have indicated that over the last 50 yr soil movement from the cultivated slope catchment to the river system is mainly in the form of bulk soil. The present investigations also

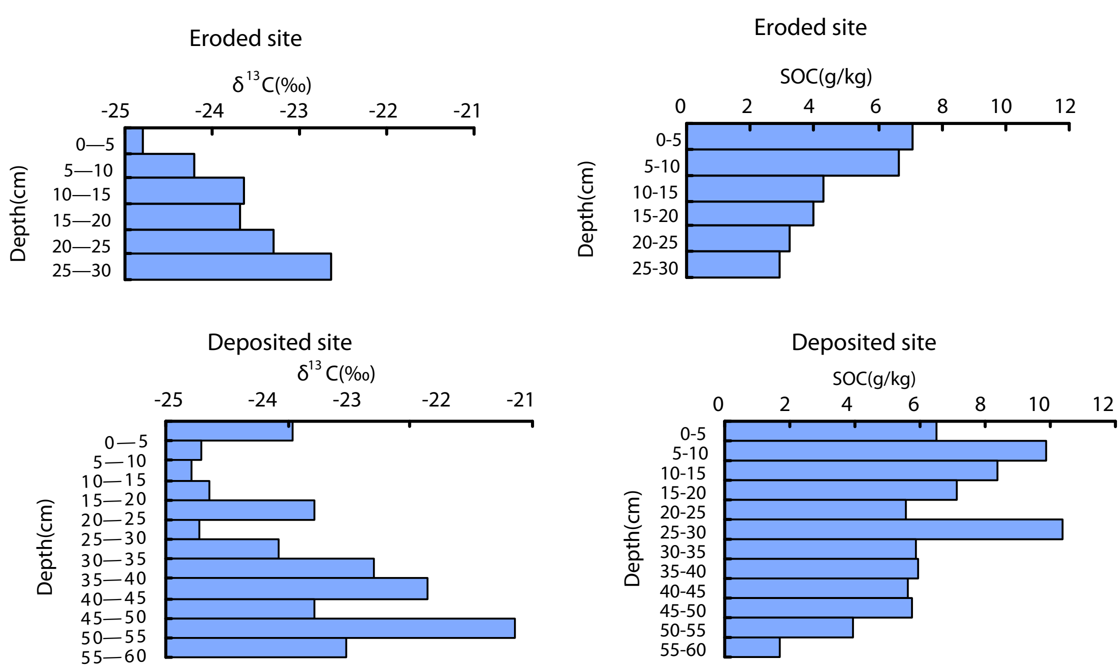

FIGURE 5. Profile distributions of SOC and $\delta^{13}C$ at eroded and deposited sites.

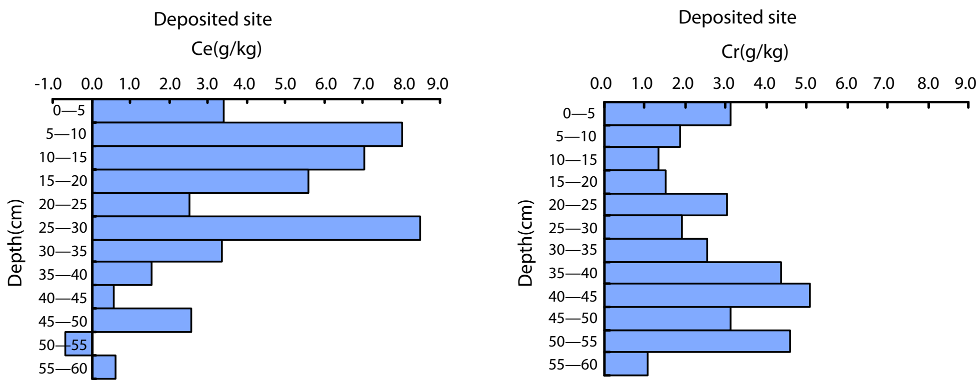

FIGURE 6. Profile distribution of SOC sources identified using δ^{13}C.

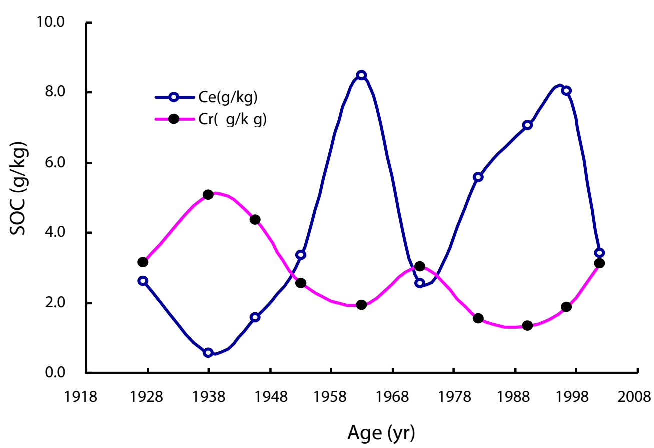

FIGURE 7. Reconstructed changes since 1918 in SOC sources in deposited profile at the lower field boundary of the cultivated slope. Ce — SOC eroded soil C from upland; Cr (g/kg); Cr — C input by crop roots (g/kg).

showed that soil accumulation at the lower field boundary was dominated by tillage erosion processes. It is therefore possible to use the reconstructed changes in SOC sources in the deposited profile (Figure 7) to explain the roles of water erosion and intensive tillage processes in bringing about upland degradation during the period 1927–2008. For example, in the period 1927–1954, the mean value of Ce in the deposited profile (2.03 g/kg) was lower than the values for SOC below the plough layer of 15 cm (these ranged from 2.92 to 3.97 g/kg with a mean of 3.37 g/kg). This suggests that sheet erosion was mainly responsible for land degradation of the cultivated slope catchment between 1927 and1965. By contrast, over the period 1954–2008, the mean value of Ce in the deposited profile (5.85 g/kg) was almost the same as the values for SOC within the plough layer of 0–15 cm (range 4.28–7.11 g/kg with a mean of 6.00 g/kg). This suggests that bulk soil translocation processes arising from intensive tillage were responsible for the land degradation of the cultivated slope catchment between 1954 and 2008.

Changes in upland crop productivity could be determined from the Cr values derived from the deposited profile (Figure 7). In the years before 1954 these were much higher than subsequently. Thus, during the period between 1927 and 1954, Cr values ranged from 2.54 to 5.07 g/kg with a mean of 3.78 g/kg, while for the period between 1954 and 2008, these ranged from 1.32 to 3.11 g/kg with a mean of 2.13 g/kg, i.e. a 44 percent decrease compared with the earlier period. This indicates that significant agricultural land degradation resulted from intensive tillage activities over the last 50 years.

CONCLUSIONS

By using ^{137}Cs and ^{210}Pb$_{ex}$ integrated with terrain attributes, models were established for slope-catchment evaluation of SOC and TN stocks covering a time period of 50–100 yr. These models had a very high degree of accuracy for quantifying changes in SOC and TN stocks in a cultivated slope catchment.

By using FRN profile dating in combination with natural δ^{13}C tracer, the roles of water and intensive tillage erosion in bringing about upland degradation over the past 100 yr could be explained.

Future work will be directed towards validating and up-scaling the models for the entire Loess Plateau and for testing the suitability of combining FRN and δ^{13}C methodologies for understanding upland degradation induced by soil erosion on an area-wide scale.

ACKNOWLEDGEMENTS

This study was supported by the National Natural Science Foundation of China (grant nos. 41171231 and 31000944), and by the International Atomic Energy Agency (IAEA) under Research Contract nos. 15479, 15938, 16290 and 16941.

REFERENCES

IAEA. 2008. Report of an FAO/IAEA Consultants Meeting on integrated approaches for the assessment of land use impacts on soil loss/sediment production and related environmental problems, 5–7 November 2007, Vienna. (http://www-naweb.iaea.org/nafa/swmn/public/snl-30-2.pdf).

Lal, R. 2007. Farming carbon. *Soil Till. Res.*, 96: 1–5.

Latta, R. & Lal, R. eds. 1999. *Soil quality and soil erosion*. R. Latta & R. Lal eds. Boca Raton, London, New York & Washington DC, CRC Press. 352 pp.

Li, Y. & Lindstrom, M.J. 2001. Evaluating soil quality-soil redistribution relationship on terraces and steep hillslope. *Soil Sci. Soc. Am. J.*, 65:1500–1508.

Li, Y., Poesen, J. & Valentin, C. 2004a. Gully erosion under global change. China, Sichuan Science & Technology Press.

Li, Y., Tian, G., Lindstrom, M. & Bork, H.R. 2004b. Variation of surface soil quality parameters by intensive donkey-drawn tillage on steep slope. *Soil Sci. Soc.* Am. J., 68: 907–913.

Li, Y., Zhang, Q.W., Reicosky, D.C., Bai, L.Y., Lindstrom, M. & Li, L. 2006. Using ^{137}Cs and ^{210}Pbex for quantifying soil organic carbon redistribution affected by intensive tillage on steep slopes. *Soil Till. Res*., 86: 176–184.

Li, Y., Zhang, Q.W., Reicosky, D.C., Lindstrom, M.J., Bai, L.Y. & Li, L. 2007. Changes in soil organic carbon induced by tillage and water erosion on a steep cultivated hillslope in the Chinese Loess Plateau from 1898–1954 and 1954–1998. *J. Geophys. Res.*, 112: G01021, 10pp.

Nearing, M.A., Pruski, F.F. & O' Neal, M.R. 2004. Expected climate change impacts on soil erosion rates: a review. *J. Soil Water Conserv*., 59: 43–50.

Pennock, D.J. 2005. Precision conservation for co-management of carbon and nitrogen on the Canadian prairies. *J. Soil Water Conserv*., 60: 396–401.

Poesen, J., Boardman, J., Wilcox, B. & Valentin, C. 1996. Water erosion monitoring and experimentation for global change studies. *J. Soil Water Conserv*., 51: 386–390.

Wallbrink, P.J. 2004. Quantifying the erosion processes and land uses which dominate fine sediment supply to Moreton Bay, South Queensland, Australia. *J. Environ. Radioact*., 76: 67–80.

Walling, D.E., Collins, A.L. & Sichingabula, H.M. 2003. Using unsupported lead-210 measurements to investigate soil erosion and sediment delivery in a small Zambian catchment. *Geomorph.*, 52: 193–213.

Walling, D.E., He, Q. & Appleby, P.G. 1997. Conversion models for use in soil-erosion, soil-redistribution and sedimentation investigations. *In* F. Zapata., ed. *Handbook for the assessment of soil erosion and sedimentation using environmental radionuclides*, pp. 111–164. New York, Springer.

Determining Sources of Soil Erosion using Compound Specific Isotope Analysis: Application in a Rural Australian Catchment

G.J. Hancock[1,*] and A.T. Revill[2]

SUMMARY

Compound Specific Isotope Analysis (CSIA) was used to assess the ability of the delta carbon-13 ($\delta^{13}C$) signature of fatty acid compounds to discriminate erosion sources in the Logan-Albert catchment, a rural river catchment on the Australian east coast. The study augmented a previous sediment tracing study using fallout radionuclides and major/minor element geochemistry. Soil samples representing important erosion sources within the catchment were collected and the $\delta^{13}C$ values of bulk carbon and various fatty acids determined. It is found that surface soil from forest, pasture and cultivated land uses are well discriminated using CSIA. Furthermore, sub-surface soil sources associated with channel bank erosion and exposed subsoils (gullies and hillslope scalds) occurring specifically in the mid-western Logan catchment could also be discriminated. Sediment samples deposited during a high flow event in January 2008 were also collected and analysed, and the IsoSource mixing model was used to determine erosion sources contributing sediment. The results were compared with results obtained using other sediment tracers. For the lower Logan River, the CSIA tracing results are consistent with fallout radionuclide and element geochemistry tracing, with channel bank erosion being confirmed as the major sediment source. Moreover CSIA has quantified the significant contribution of exposed subsoils originating on hillslopes and drainage lines from the mid-western region of the Logan catchment. In the Albert River catchment about 40 percent of sediment comes from forest land use, although more than half of this may come from sub-surface sources. These results have demonstrated that CSIA has the potential to enhance significantly the ability of sediment tracing studies to determine the extent to which different land uses and erosion processes are contributing eroded soil to rivers, thus testing and validating model predictions and calibration of model parameters.

Key words: *compound specific isotope analysis (CSIA), $\delta^{13}C$, fatty acids, soil erosion, sediment tracing, land use.*

INTRODUCTION

Determining the proportions of eroded soils contributing sediment to rivers and streams using physical and chemical characteristics can be a direct and cost-effective way of directing erosion control actions. The method is particularly powerful when used in conjunction with catchment modelling and in-stream monitoring of suspended sediment loads to determine catchment sediment budgets (Hancock *et al*., 2007; Rustomji *et al*., 2008). The soil characteristics, or tracers, can take many forms: soil colour, geochemistry, stable isotope composition and mineralogy have been successfully employed to trace the spatial and/or geomorphological origin of sediment (e.g. Douglas *et al*., 1995 and 2003; Walling 2005). Fallout radionuclides have also been found to be valuable tracers of surface soil erosion (Olley *et al*., 1993; Wallbrink *et al*., 1996 and 1998; Walling and Woodward, 2002; Walling 2005). In particular, anthropogenic caesium-137 (^{137}Cs) and naturally occurring fallout lead-210 (^{210}Pb) label the upper few cm of surface soil globally, providing an indication of whether erosion of surface or sub-surface soils are contributing to river sediment.

While these tracers, either separately or in combination, provide important erosion source information, catchment managers are often called to make soil conservation decisions on the basis of land use. Since different land uses can span geological boundaries and can lead to erosion of both surface and sub-surface soil there is no guarantee that traditional tracers can distinguish soils originating from (for example) forest, pasture or crop cultivation. Recent studies have utilized the stable isotope signature of a range of organic compounds in sediments to determine carbon (C) sources delivered to rivers and near-shore waters (e.g. Prahl *et al*., 1994; Cook *et al*., 2004; Seki *et al*., 2010). Subsequently, the Compound Specific Isotope Analysis (CSIA) technique has been used to determine sediment sources using the delta carbon-13 ($\delta^{13}C$) signature of C compounds bound to the sediment particles as a tracer of their source. Compound Specific Isotope Analysis has successfully distinguished terrestrial and estuarine sediments (Hu *et al*., 2006) and has been applied recently in New Zealand to distinguish catchment sources of eroded surface soil under different land uses (Gibbs, 2008), and in Britain to distinguish the effect of different crops on downstream stream sediment load (Blake *et al*., 2012).

Compound specific $\delta^{13}C$ measurements as sediment tracers

The CSIA technique measures the $\delta^{13}C$ isotope signature of specific organic compounds associated with the organic matter bound to the soil/sediment. In this paper we investigate the use of fatty acids (FAs), organic compounds which are produced by plants and soil organisms. Due to their polar nature FAs are easily leached from the plant and can become tightly bound to soil particles (Thurman, 1985). It is proposed that different land uses will provide distinctive CSIA signatures due to the different types of vegetation and com-

1 CSIRO Land and Water, GPO Box 1666 Canberra, ACT 2601, Australia
2 CSIRO Marine and Atmospheric Research, GPO Box 1538, Hobart, Tasmania 7001, Australia
* E-mail address of corresponding author: Gary.Hancock@csiro.au

L.K. Heng, K. Sakadevan, G. Dercon and M.L. Nguyen (eds), Proceedings — International Symposium on Managing Soils for Food Security and Climate Change Adaptation and Mitigation. Food and Agriculture Organization of the United Nations, Rome, 2014: 133–141

munity structure which, on decomposition, make up the organic matter bound to soil particles (Chikaraishi and Naraoka, 2003; Gibbs, 2008). It is well known that various plants (e.g. grasses, shrubs, trees etc.) synthesize ^{13}C differently, leading to different δ^{13}C values for the vegetation of those plants (e.g. Peterson and Fry, 1987; Lamb *et al.*, 2006). On decomposition of this vegetation the δ^{13}C values of the various C compounds are retained and transferred to the soil. Thus, different plants (e.g. "C3" and "C4" plants) associated with different land uses may allow the identification of erosion sources contributing sediment to rivers and estuaries. Despite continued degradation of FAs over time the isotopic signature is retained (Blessing, Jochmann and Schmidt, 2008).

While CSIA shows promise as a tracing technique of surface soils, it has not yet had widespread application, nor has it been applied to distinguish sub-surface erosion sources. In the Australian context the method has the potential to provide valuable information on erosion in rural areas such as those typified by coastal catchments along the eastern and southern coasts. The technique could be particularly relevant in the coastal catchments of central and northern Queensland where debate is occurring on the major erosion sources contributing sediment and nutrient to the Great Barrier Reef (GBR) lagoon. In particular, the relative contributions of human-induced erosion associated with various agricultural practices (cropping, irrigated and native pasture) are being compared with "natural" fluxes from native forest. While fallout tracers can identify surface soil erosion, they cannot directly determine the contributions of the different land uses, and estimates of erosion sources are often based on un-validated model estimates. Moreover, it is known that in many of these catchments erosion of soil from sub-surface sources dominates sediment fluxes. Fallout tracers have only a limited ability to distinguish these sources; for example both channel bank and gully erosion have little or no ^{137}Cs and excess ^{210}Pb. Neither, in many cases, does soil geochemistry provide discrimination, since soil geochemistry is largely controlled by the geology of the source rocks from which the soils are derived, and erosion processes are not always related to catchment geology. However, C sources of different sub-surface soils may differ, either as result of historical changes reflected by a contrast between B-horizon and A-horizon soils, or by the source of C being delivered to floodplains and channel banks. We therefore suggest that CSIA has the potential to distinguish different sources of sub-surface soils.

This paper describes work designed to assess the ability of the CSIA technique to discriminate eroded soil (and associated C) sources in a rural Australian catchment. We report the fatty acid CSIA characterization and discrimination of soil and sediment samples from the Logan and Albert River catchments. These two adjoining sub-tropical catchments are considered representative of many Australian rural eastern coastal catchments, both in regard to land use and the environmental concerns associated with eutrofication of estuarine waters. The results here may also have application to tropical catchments further north making up the GBR lagoon catchment. The Logan and Albert catchments were the subject of a study using traditional radionuclide and major/minor element tracers (Hancock and Caitcheon, 2010), the results of which provided new information on sediment sources and erosion processes. However these tracers could not discriminate between different land uses within the same geological province, in particular between the proportions of sediment originating from pasture, forest and cultivated regions. The different sub-surface soil sources were also not easily distinguished from each other. This study investigates the ability of CSIA to differentiate surface soils associated with specific land uses, particularly native forest and pasture. We also assess its ability to distinguish sub-surface soils, such as channel bank, and hillslope scalds and incised drainage lines. The proportioning of sources using CSIA is compared with previously published results using other tracers.

Study site

The catchment of the Logan and Albert Rivers comprises an area of 3 860 km^2 and is mainly rural. Three main water courses deliver water to the estuary: the Logan River with its tributary Teviot Brook, and the Albert River (Figure 1). Water flow is from the uplands in the south, to the estuary in the north. The dominant land uses are grazing (cattle), native forest, cropping and rural residential. The catchment headwaters are forested conservation zones (Lamington and Mt. Barney National Parks) and are in relatively pristine condition, but the catchment has been extensively cleared elsewhere for cropping, grazing and dairying. Grazing occurs in the middle and upper parts of the catchment and includes improved pasture and partially cleared grassed woodlands. Cultivation (cropping) occurs mainly along the river flats and floodplains of the major tributaries.

The Logan estuary is situated in the south of Moreton Bay and is characterized by high nutrient levels and turbidity. The ecological health of streams of the Logan catchment and the estuary is considered poor. The health of the Albert River is considered to be good, but its estuary is connected to the Logan estuary and it has a similarly poor ecological health ranking.

METHODS

Sample collection

Catchment soil samples and sediment samples for CSIA were collected from January 18–21, 2010. The sampling process was structured so that samples were collected representing the various potential sources of eroding soil contributing sediment transported from the catchment. Surface soil samples were obtained by collecting the top ~10 mm of soils from hillslopes and cultivated paddocks. Only the "mobile" or "potentially mobile" component of the surface soil was collected. These soils are defined as surface soil that had clearly been transported down-slope after recent rain, or was loose enough such that transport by overland flow was considered likely to occur during the next major rainfall event.

Soil samples representing distinct erosion sources were obtained by combining between three and five samples collected from hillslopes and paddocks with similar characteristics within a few km of each other. All combined samples represented the same major land use, and showed similar slope and vegetation cover. At each site 30–50 individual "grab" samples were taken over an area of at least 10 000 m^2. These were combined into a single sample. Thus each representative surface soil sample contained 100–200 individual grab samples collected along 10–20 km transects. Soil sampling locations are shown in Figure 1.

The surface soil samples were collected across the main tributaries of the Logan-Albert catchment, and included soils from the three main land uses:

- National Park and State Forest hillslopes.
- Permanent (uncultivated) pasture grazed by cattle for beef and dairy.
- Cultivated soils currently used for a range of crops planted and harvested throughout the year on a rotational basis (e.g. corn, sorghum, lucerne and various vegetables).

As noted above, pasture generally comprised gently sloping grassed hillslopes, either completely or partly cleared of woody vegetation. Protected forest hillslopes typically were steeper with extensive canopy cover, with little or no grass. Cultivated soils occur on or

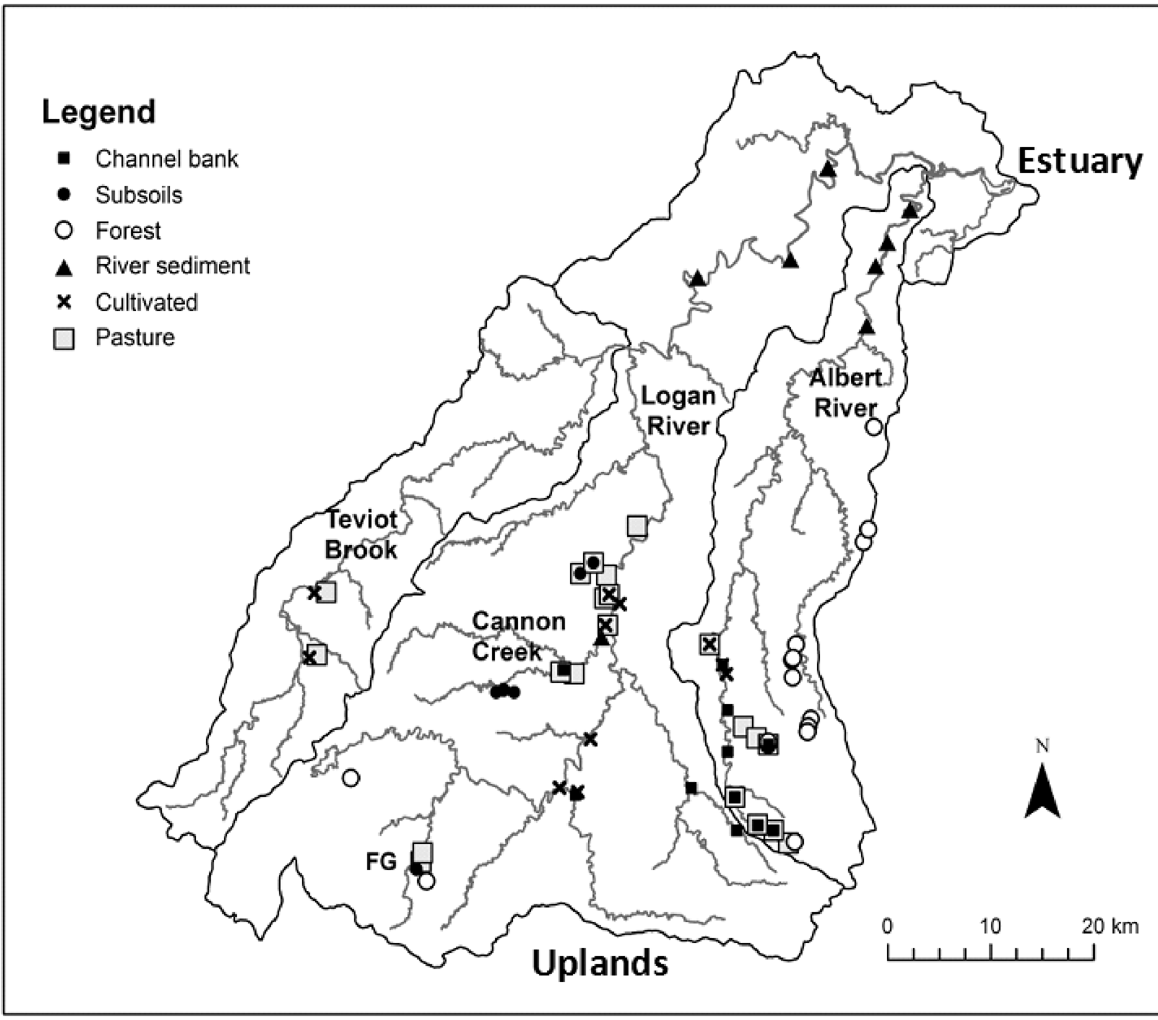

FIGURE 1. Map of the Logan-Albert catchment and locations of soil and sediment samples. Catchment boundaries are given by the dark solid line. Note that the sediment samples (closed triangles) for the lower Logan and Albert rivers were combined into a single sample for CSIA analysis. The forest gully sample is labelled as FG.

near river floodplains. Similarly, subsoil samples were collected from gully and scald sites that were clearly identifiable as being actively eroding. Again, only the mobile fraction was collected. Channel bank samples were collected from three sites along an approximately 5 km river reach and combined for analysis. Similar to hillslope sampling, multiple grab samples were collected at each site. Each eroding channel bank profile was sampled by scraping a thin layer (< 10 mm) of soil from the exposed vertical bank face. Because the whole vertical face was sampled a small component of surface soil or overbank sediment deposits from the top of the bank was included in the sample.

In addition to soil samples collected in 2010, a selection of river sediment samples collected in the January 2008 field trip immediately after the flood event were also analysed by CSIA (Figure 1, closed triangles). These samples represent deposits of fine sediment associated with the January 2008 flood event and were stored frozen. For the lower Logan and Albert rivers, sediment from four sample sites were combined into a single sample representative of the lower reaches of the River, and analysed by CSIA. A single sample from Cannon Creek, in the mid Logan catchment was also analysed.

Sample preparation and analysis

Representative soil samples were obtained by sub-sampling the bulk samples to obtain a single sample of ~100 g dry weight. Stored sediment samples were thawed and sieved in the same way. In the laboratory samples were extracted with n-hexane. The lipids were recovered and made up to a known volume with dichloromethane. Fatty acids were recovered from the aqueous fraction of the saponified mixture after the addition of 1 mL of HCl and converted to their methyl esters prior to analysis by treatment with acidified methanol (Christie, 1982).

Gas chromatography was performed using a Varian 3 800 equipped with a cross-linked 5 percent phenyl-methyl silicone (HP5) fused-silica capillary column using hydrogen as the carrier gas. Fatty acid fractions were analysed using a flame ionization detector, with the C_{23} fatty acid methyl ester as the injection standard. Compound specific isotope analysis was performed using a Thermo GC coupled to a Finnigan Mat Delta S isotope ratio mass spectrometer operating in continuous flow mode with helium as the carrier gas. Stable isotope values were corrected for the C added during methylation; the correction amounted to less than 1‰. Analytical precision is estimated at better than 0.5‰.

RESULTS AND DISCUSSION

Characterizing soil sources

Soil samples were grouped into five erosion sources: surface soils from hillslopes under the two major land uses (forest and pasture); the two major sub-surface erosion sources (in-stream channel banks and hillslope scalds) delivering soil derived from beneath the surficial soil layer (the A-horizon, or the upper ~10–15 cm). These two sub-surface soil sources are referred to separately as "channel bank" and "subsoils" the latter relating to hillslope scalds and incised drainage lines clearly seen in the mid-western region of the Logan catchment. The fifth erosion source is cultivated paddocks, which, due to the tilling process will contain both surface and sub-surface soils.

Mean concentrations for each fatty acid (FA, units of µg/g) and bulk C (units of % dry weight) for the five soil sources are shown in Figure 2. The FA concentrations range from about 10 µg/g to less than 0.1 µg/g with the highest concentrations shown by the 16 and 18 C-chain derivatives (C16:1w7, C16:0, C18:1w, C18:0). The highest soil source FA concentrations are shown by forest soils and the lowest by hillslope subsoils, the latter being derived primarily from B-horizon soils. This trend closely matches the bulk C concentrations, with forest soils having an average 7.7 percent C (dry weight), and subsoils 0.6 percent C. The FA concentration trend closely matches the bulk C trend for all soil sources.

The $\delta^{13}C$ FA data are summarized in Table 1, which gives mean $\delta^{13}C$ values for all FAs and bulk C for each soil source. Also shown is the standard error (±1 sigma) on the mean. Not all soils yielded meaningful $\delta^{13}C$ values for all the FAs, especially the higher chain FAs derived from soils with low C content. Overall, there is good separation between the FA $\delta^{13}C$ values for pasture and forest soils, with pasture generally showing the highest $\delta^{13}C$ (mean value range –13 to –26‰) and the forest soil showing the lowest (mean value range –22 to –36‰). These trends are also shown by the bulk C $\delta^{13}C$ measurements, with mean bulk C $\delta^{13}C$ values for pasture soils (–15.3‰) and forest soils (–24.4‰) showing the greatest difference. These results are consistent with previous work on bulk soil C, where bulk $\delta^{13}C$ values for warm-climate grasses (C4 plants) were found to be less negative and well separated from those of shrubs and trees (C3 plants) (Fry and Sherr, 1984; Peterson and Fry, 1987). The FA and bulk $\delta^{13}C$ values for the other source soils lie between the pasture and forest values.

The CSIA discrimination between soil sources was quantified using the statistical t-test to compare the significance of differences between pairs of soil sources for each fatty acid. The results of the test are expressed as the parameter T_{ab}, given by the difference between the FA mean $\delta^{13}C$ values of source soils a and b, divided by the standard error of that difference; i.e.

$$T_{ab} = \frac{\left|\overline{X}_a - \overline{X}_b\right|}{\sqrt{(\sigma_a^2 + \sigma_b^2)}} \qquad (1)$$

where $\overline{X}_a$ and $\overline{X}_b$ represent the mean $\delta^{13}C$ values of each FA and σ_a, σ_b represent the standard errors on the mean.

T_{ab} values for the 10 soil source pairs are shown in Table 2. Some fatty acids (C20:0 to C30:0) are not included because subsoil data are missing. Values of T_{ab} can range from 0 (no discrimination) to >5 (excellent) and soil sources with T_{ab} values of 5 and above (equivalent to a statistical t-test with $p < 0.01$) were selected for the mixing model. These values are summarized in Table 2. Excellent pasture:forest discrimination is seen and both pasture and forest are well discriminated from the other sources. The two subsoil sources (channel bank and subsoils) and cultivated soils show a lesser degree

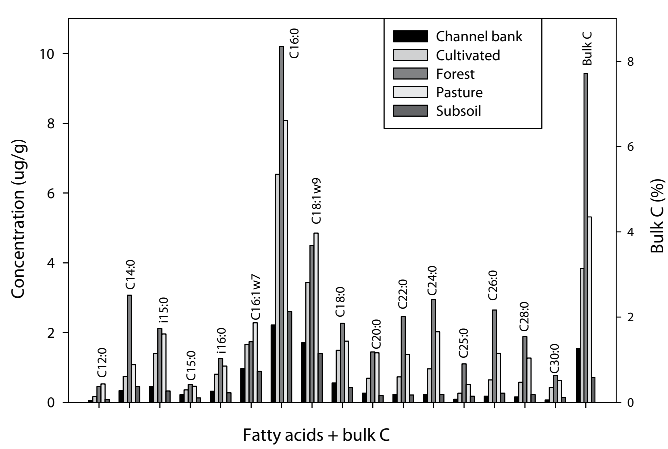

FIGURE 2. Fatty acid concentrations for the five soil sources. Also shown is % bulk C (dry weight). Note separate y-axis for % bulk C.

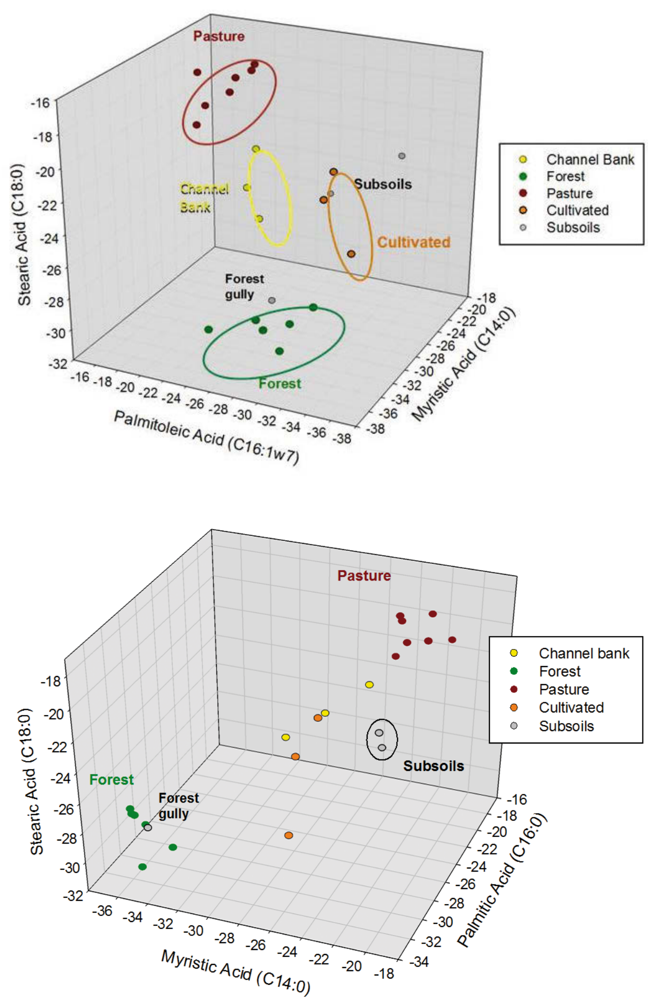

FIGURE 3. A (Top): 3-D plot of soil sources and three FAs showing the separation between sources. B (Bottom): The same plot rotated to the left showing the separation of the two west-Logan subsoil sources (circled). The third subsoil sample, the forest gully sample from Mt Barney NP, plots with the forest samples.

TABLE 1. Mean % C and $\delta^{13}C$ (‰) for FAs and bulk C for each soil source. Note the forest gully sample is excluded from the subsoil mean value. Standard error (SE) is given (±1σ), with the number of measurements given in brackets. Note that the number of measurements varies for each source due to the low concentrations of some FAs not allowing a meaningful measure of $\delta^{13}C$.

	Channel Bank		Cultivated		Forest		Pasture		Subsoils	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
C (%)	1.26	0.24 (3)	3.14	0.34 (3)	7.72	1.23 (6)	4.35	0.49 (7)	0.59	0.10 (2)
Bulk $d^{13}C$	–21.10	0.33(3)	–18.85	0.69 (3)	–24.42	0.79 (6)	–15.30	0.56 (7)	–20.64	0.70 (2)
C12:0	–22.02	1.26 (3)	–22.59	0.49 (4)	–31.44	1.52 (6)	–21.10	0.54 (7)	–20.66	0.72 (2)
C14:0	–26.86	1.21 (3)	–26.24	0.23 (4)	–35.25	0.51 (6)	–22.46	0.43 (7)	–22.51	0.47 (2)
i15:0	–20.64	1.79 (3)	–19.87	0.57 (4)	–26.34	0.47 (6)	–15.40	0.44 (7)	–19.60	0.06 (2)
a15:0	–18.71	1.09 (3)	–20.59	0.90 (4)	–22.10	0.77 (6)	–13.20	0.49 (7)	–19.28	0.49 (2)
C15:0	–24.01	2.32 (3)	–22.03	0.69 (4)	–32.82	1.72 (6)	–19.86	1.46 (7)	nm	
i16:0	–24.92	1.90 (3)	–20.87	0.45 (4)	–30.16	0.71 (6)	–17.64	0.25 (7)	–19.71	1.00 (2)
C16:1w7	–23.23	0.83 (3)	–30.05	0.62 (4)	–29.33	1.09 (6)	–17.96	0.85 (7)	–30.64	2.80 (2)
C16:0	–23.76	0.82 (3)	–29.01	1.30 (4)	–31.78	0.46 (6)	–20.70	0.78 (7)	–26.05	1.56 (2)
18 comb	–23.78	0.79 (3)	–24.19	1.23 (4)	–26.89	0.13 (6)	–18.12	0.56 (7)	–22.96	0.95 (2)
C18:0	–23.41	0.80 (3)	–23.95	1.10 (4)	–28.47	0.49 (6)	–19.11	0.55 (7)	–23.24	1.42 (2)
C20:0	–23.77	3.40 (2)	–23.97	1.01 (4)	–30.73	0.94 (6)	–20.79	0.36 (7)	–18.54	(1)
C22:0	–29.22	1.99 (2)	–27.17	0.73 (4)	–32.92	0.28 (6)	–23.46	0.55 (6)	nm	
C24:0	–31.65	(1)	–26.93	1.24 (4)	–32.29	0.40 (6)	–23.05	0.66 (6)	nm	

nm — not able to measure

TABLE 2. Discrimination between soil sources by fatty acid CSIA and bulk $\delta^{13}C$ as measured by T_{ab} (see text for details). Values of T_{ab} greater than 5 are shaded. Fatty acids with missing subsoil data (C20:0 to C30:0) have not been included. The FAs selected for mixing model analysis are also indicated by shading of the column headings.

Soil source pairs	Bulk $\delta^{13}C$	C12:0	C14:0	i15:0	a15:0	i16:0	C16:1w7	C16:0	18 comb	C18:0
Pasture:Forest	9.4	6.4	19.1	17.0	9.8	16.6	8.2	12.2	15.2	12.6
Pasture:Cultivated	4.0	2.0	7.7	6.2	7.2	6.3	11.5	5.5	4.5	3.9
Pasture:Channel Bank	8.9	0.7	3.4	2.8	4.6	3.8	4.5	2.7	5.8	4.4
Pasture:Subsoils	5.9	0.5	0.1	9.5	8.7	2.0	4.3	3.1	4.4	2.7
Forest:Cultivated	5.3	5.5	16.1	8.7	1.3	11.0	0.6	2.0	2.2	3.8
Forest:Channel Bank	3.9	4.8	6.4	3.1	2.5	2.6	4.5	8.5	3.9	5.4
Forest:Subsoils	3.6	6.4	18.3	14.1	3.1	8.5	0.4	3.5	4.1	3.5
Channel Bank:Cultivated	2.9	0.4	0.5	0.4	1.3	2.1	6.6	3.4	0.3	0.4
Channel Bank:Subsoils	0.6	0.9	3.4	0.6	0.5	2.4	2.5	1.3	0.7	0.1
Cultivated:Subsoils	1.8	2.2	7.1	0.5	1.3	1.1	0.2	1.5	0.8	0.4

of discrimination, although at least one FA shows good discrimination for each source pair. Based on this analysis the following FAs were selected to provide the best overall discrimination for all soil sources: myristic acid (C14:0), palmitoleic acid (C16:1w7), branched-chain iso$C_{15:0}$ (i15:0), and stearic acid (C18:0). Palmitic acid (C16:0) was substituted for (C16:1w7) for some IsoSource runs. Following the suggestion of Gibbs (personal communication), the fifth input term (tracer) used in IsoSource was $\delta^{13}C$ of the bulk (whole) sample as it makes up the largest mass component of the soil (1 000-fold greater than the FA signatures).

The discrimination of sources by isotopic composition of three members of the FA suite can be assessed visually using a 3-D plot (Figure 3). Good separation can be seen between the three land uses (pasture, forest and cultivation), as well as river channel banks. Rotation of the plot and substituting palmitic acid (C16:0) for palmitoleic acid (C16:1w7) (Figure 3B) allows separation of the two subsoil samples from the western mid-Logan catchment. It is interesting to note that the subsoil sample collected from a gullied drainage line in National Park forest plots close to surface soil samples from forested areas.

Determining sediment sources

Mixing models are primarily used to apportion sources to sediment comprising a mixture of soils. As suggested by Gibbs (2008) we have used the isotopic mixing model IsoSource (Phillips and Gregg, 2003) to estimate soil sources for sediment samples collected at various locations in the river network. The model calculates the feasible solutions within a given tolerance (e.g. ± 1‰) of the tracer concentrations of the target mixture. Results are displayed as a frequency

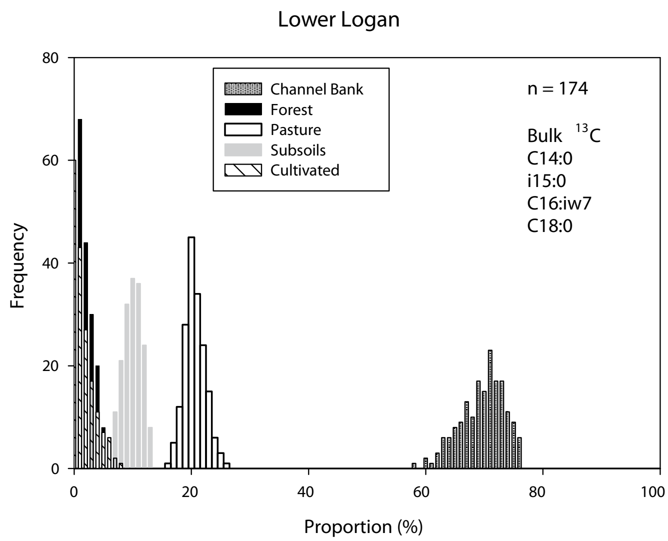

FIGURE 4. IsoSource frequency plot for the proportions of soil sources contributing to sediment from lower Logan River. The five tracers used are listed; the number of feasible solutions, n, is 174.

distribution of all feasible solutions. Whereas each feasible solution is equally possible, the most frequently occurring solution (statistical mean) provides a best estimate and the range of feasible solutions represents the level of uncertainty.

Because the CSIA method traces C rather than the soil itself, the results given by the mixing model need to be adjusted to account for the different amounts of each FA in each of the soil sources. This is done by using the % C content of each source, as described by Gibbs (2008), i.e.

$$\%Soilsource_n = \frac{(P_n / C_n)}{\sum_n (P_n / C_n)} \times 100 \qquad (2)$$

where P_n (%) is IsoSource mean solution for source soil n; and C_n is % C of the soil.

An example of the output of soil source apportioning by IsoSource is shown in Figure 4. Five soil sources are considered. The target sediment mixture is a combined sediment sample collected from the lower Logan River in January 2008, representing Logan River sediment downstream from its junction with Teviot Brook to the beginning of the Logan estuary, a distance of ~40 km (Figure 1). Four sediment samples collected along this reach were combined into a single sample and analysed by CSIA. The frequency distribution plot in Figure 4 indicates that the FA and C contents of lower Logan sediment are sourced mainly from channel bank soil (68 ± 4 percent), with lesser amounts from pasture soil (20 ± 2 percent) and subsoils (9 ± 2 percent). Only minor amounts of cultivated and forest soil are indicated, with no valid solutions being seen above 8 percent for either, and most of the solutions being less than 3 percent. All soil sources are well constrained.

Table 3 summarizes the IsoSource mean proportions and uncertainties for each soil source contributing C to the lower Logan sediment. Also shown are the proportions of each soil source after application of Equation 2 to correct for the percent C content of each source. The relatively high C content of the pasture soil and the relatively low content of subsoils has seen their proportions decreased and increased respectively (pasture = 6 ± 1 percent; subsoils 20 ± 4 percent). The channel bank proportion is almost unchanged.

TABLE 3. Apportionment of FA (C) sources as determined from δ^{13}C values using IsoSource (column 2). Soil sources in column 3 are then determined using bulk C values and Equation 2

	Bulk C (%)	Nominal C sources (%)	Soil sources (%)
Lower Logan			
Channel bank	1.26	68 ± 4	72 ± 4
Cultivated	3.14	2 ± 2	1 ± 1
Forest	7.72	1 ± 1	< 1
Pasture	4.35	20 ± 2	6 ± 2
Subsoils	0.59	9 ± 2	20 ± 4
Cannon Creek			
Channel bank	1.26	22 ± 12	17 ± 10
Forest	7.72	11 ± 5	1 ± 1
Pasture	4.35	20 ± 6	5 ± 1
Subsoils	0.59	47 ± 3	77 ± 6
Lower Albert			
Channel bank	1.26	14 ± 2	48 ± 8
Cultivated	3.14	7 ± 2	9 ± 3
Forest	7.72	78 ± 1	42 ± 1
Pasture	4.35	1 ± 1	< 2

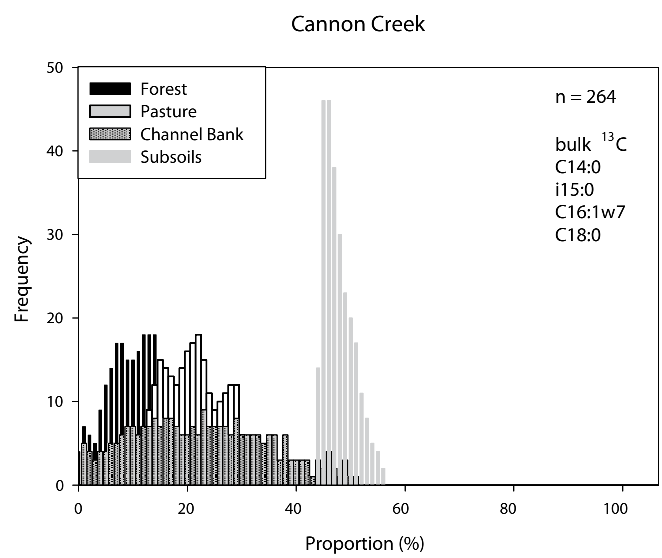

FIGURE 5. IsoSource frequency plot for the proportions of soil sources contributing to sediment to Cannon Creek. Only four soil sources are considered (cultivated soils are excluded); n = 264.

It is interesting to consider the role of the mid-western region of the mid-Logan catchment in delivering subsoils to the lower Logan. Geochemical tracing by Hancock and Caitcheon (2010) indicated that a significant proportion (up to 25 percent) of sediment delivered to the Logan River in the middle catchment region is eroded from soils associated with the Marburg geological formation. Surface tracers indicate that subsoil erosion is significant in this region. These eroded subsoils are almost certainly coming from creeks draining the mid-western region of the Logan catchment, all of which drain soils from the Marburg geological province. Figure 5 shows the IsoSource output for 2008 flood sediment collected from one of these

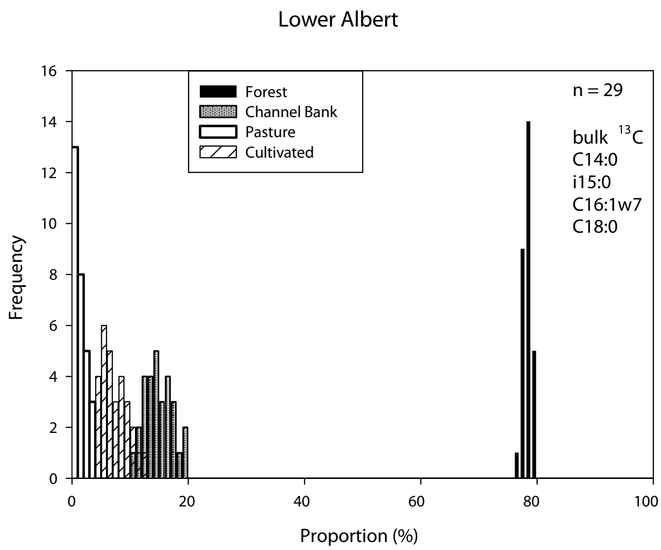

FIGURE 6. IsoSource frequency plot for the proportions of soil sources contributing to sediment from lower Albert River. Only four soil sources are considered (subsoils are excluded); (n = 29).

creeks, Cannon Creek. Crop areas are minimal in this catchment and cultivated soils were not considered as a source term.

While the resolution of minor sources is poor with the reliability of the minor source predictions being reflected by their uncertainties, IsoSource predicts a high likelihood of major input from hillslope subsoil C (range 42 to 62 percent). This is despite the low C and FA content of this source. When corrected for soil C content the model predicts that 77 ± 6 percent of Cannon Creek sediment is sourced from eroding subsoils (Table 3).

Table 4 compares the results of all tracer techniques (surface tracers, geochemistry and CSIA) for the lower Logan and Cannon C sediment samples. Fallout radionuclides are assumed to have been mostly derived from surface forest and pasture soils since they are the dominant land uses on an areal basis. The remaining sediment is assumed to come from the sub-surface sources of channel bank and subsoils (gullies) as indicated by catchment modelling results (Caitcheon *et al*., 2001). Cultivated soils could not be distinguished by fallout tracers. The results of all three tracer techniques are consistent, with the CSIA technique confirming the dominant role of channel banks as a sediment source to the lower Logan (50–70 percent) and the relatively minor surface soil inputs from forest and pasture. Cultivated soils are also a minor source. The CSIA results corroborate the indication given by fallout tracers that a smaller but significant input (20–30 percent) comes from hillslope subsoils in the western creeks catchments.

Another example of soil source estimation by CSIA is illustrated by the IsoSource results for the lower Albert River. In this analysis only four soil sources are considered due to the fact that hillslope subsoils were not sampled for this catchment. Figure 6 shows the IsoSource frequency graph of valid solutions. The low number of solutions (n = 29) indicates high reliability. It is concluded that forest soils dominate C inputs to the lower Albert (78 ± 1 percent). After a soil C content correction is applied soils sourced from the forest regions are still seen to be major (42 ± 1 percent, Table 3) with channel bank being identified as the other potentially significant source. A minor component of cultivated soil is indicated.

When compared with other tracer results (Table 4) the CSIA analysis confirmed channel bank as the major erosion source (~50 percent). Importantly though, CSIA in conjunction with fallout tracers indicated that not all soil from the forested region is surface soil, since the fallout radionuclide results show that no more than 20 percent can be sourced by sheet erosion of surface soils, compared with more than 40 percent predicted by CSIA. Fallout tracers indicate that subsoil sources are contributing in the Albert catchment (30 ± 10 percent, Hancock and Caitcheon, 2010). This information in combination with the observation that the single forest subsoil (gully) noted above matched the forest surface soil signal suggests that the high forest soil component estimated by CSIA is due in part to sub-surface

TABLE 4. Comparison of the results of sediment source tracing in the Logan and Albert River catchments using CSIA, fallout radionuclides and element geochemistry. Values given are proportions (%) of soil sources. Note fallout radionuclides are assumed to have been mostly derived from surface forest and pasture soils.

	Forest	Pasture	Cultivated	Channel Bank	Subsoils
Lower Logan					
Geochemistry	Lower Logan sediment contains a mixture of eastern catchment soils from the upper and mid-catchments (Lamington soils, 70%) and mid-western catchment soils (Marburg soils, 15–25%)				
CSIA	< 1	6 ± 1	<2	72 ± 4	20 ± 4
Fallout tracers	10 ± 5*		nc	50 ± 10	40 ± 10
Cannon Ck					
Geochemistry	Creek sediment is at least 90% mid-western catchment soils (Marburg)				
CSIA	1 ± 1	5 ± 1	nc	17 ± 10	77 ± 6
Fallout tracers	10 ± 5*		nc	90 ± 5	
Lower Albert					
Geochemistry	Lower Albert sediment is made up of at least 90% Lamington soils sourced from the forested part (the southern half) of the catchment.				
CSIA	42 ± 1	< 2	9 ± 3	48 ± 8	nc
Fallout tracers	15 ± 5*		nc	55 ± 10	30 ± 10

nc — source not considered in the model

*Note the surface soil proportion predicted by surface tracers is assumed to come from the combined sum of forest and pasture land uses soil sources within the forest. The erosion processes responsible are likely to include hillslope slumping and deep rilling.

CONCLUSIONS

In this study the $\delta^{13}C$ values of at least eight FAs gave sufficient discrimination to distinguish surface soil erosion from pasture and forest land uses. Bulk $\delta^{13}C$ also provided good discrimination. Although surface soil erosion was found to be a minor contributor to sediment in the Logan-Albert catchment, it is thought to be significant in tropical regions, contributing more than 50 percent of sediment to some northern rivers (e.g. Bartley, Olley and Henderson, 2004). Consequently there has been much debate whether grazed pastures or conservation forests are the source of this surface soil. The CSIA technique has the potential to enhance significantly the ability of sediment tracing studies to investigate this important issue.

Other sub-surface soil sources were also distinguished, although to a lesser extent. Nevertheless, for sediment transported in the Logan-Albert River system the CSIA technique was able to provide realistic estimates of the proportions of sub-surface soil sources such as channel banks and hillslope B-horizon soils, a result surface tracers and elemental geochemistry often fail to achieve. In combination, the three tracing techniques – surface tracers, element geochemistry and CSIA – offer the opportunity to discriminate five potentially important erosion sources, providing spatial, land-use and erosion process source information.

While cultivated soil in the Logan catchment could be discriminated from other sources, the signature associated with specific crops was not investigated. However, we suggest that where a single crop such as sugar cane has been harvested on a long-term (decadal) time scale, discrimination between soils eroded from cane fields, pasture and forest soils may prove viable. Thus CSIA may be an important technique to identify soil sources contributing sediment along the tropical and sub-tropical east Australian coast to near-shore regions and the Great Barrier Reef Lagoon.

Finally, since CSIA traces organic C attached to sediment the technique has the potential to provide improved estimates of the sustainability of surface (A-horizon) soils, an important issue in some agricultural regions (Bui, Hancock and Wilkinson, 2011). Since organic C is essential for a productive A-horizon, rapidly depleting surface soil stocks are likely to result directly in a loss of agricultural productivity, especially on hillslopes where the A-horizon is thin.

ACKNOWLEDGEMENTS

We acknowledge the help of Danny Hunt during the collection, sub-sampling and packaging of samples in January 2010. Dan Holdsworth (CSIRO Marine and Atmospheric Research) performed the CSIA analysis and provided the fatty acid and bulk C isotope data. Discussions with Max Gibbs (National Institute of Water and Atmospheric Research) aided our understanding and application of the CSIA method.

This work was funded by CSIRO Land and Water through the Capability Development Fund and was undertaken as part of the FAO/IAEA Co-ordinated Research Project "Integrated Isotopic Approaches for an Area-wide Precision Conservation to Control the Impacts of Agricultural Practices on Land Degradation and Soil Erosion".

REFERENCES

Bartley R., Olley J. & Henderson A. 2004. Sediment transfer through the fluvial system, a sediment budget for the Herbert River catchment, North Queensland, Australia. *In* V. Golosov, V. Belyaev & D. Walling, eds. *Sediment transfer through fluvial systems.* Proceedings Moscow Symposium, pp. 147–154. IAHS Publication no. 288, Wallingford, UK, IAHS Press.

Blake W.H., Ficken K.J., Taylor P., Russell M.A., Desmond E. & Walling D.E. 2012. Tracing crop-specific sediment sources in agricultural catchments. *Geomorph.*, 139: 322–329.

Blessing M., Jochmann M.A. & Schmidt T.C. 2008. Pitfalls in compound-specific isotope analysis of environmental samples. *Anal. Bioanal. Chem.*, 390: 591–603.

Bui, E.N., Hancock, G.J. & Wilkinson, S.N. 2011. "Tolerable" hillslope soil erosion rates in Australia: Linking science and policy. *Agric. Ecosyst. Environ.*, 144: 136–149.

Caitcheon G., Prosser I., Wallbrink P., Douglas G., Olley J., Hughes A., Hancock G. & Scott A. 2001. *Sediment delivery from Moreton Bay's main tributaries: A multifaceted approach to identifying sediment sources.* Proceedings 3rd Australian Stream Management Conference, Volume 1, pp. 103–107, Brisbane Queensland, August 27–29, 2001.

Chikaraishi Y. & Naraoka H. 2003. Compound specific $\delta D–\delta^{13}C$ analyses of n-alkanes extracted from terrestrial and aquatic plants. *Phytochem.*, 63: 361–371.

Christie, W.W. 1982. *Lipid analysis, 2nd Edition*. Oxford, UK, Pergamon Press.

Cook P.L.M., Revill A.T., Clementson L.A. & Volkman J.K. 2004. Carbon and nitrogen cycling on intertidal mudflats of a temperate Australian estuary. III sources of organic matter. *Mar. Ecol.–Prog. Series*, 280: 55–72.

Douglas G.B., Gray C.M., Hart B.T. & Beckett R. 1995. A strontium isotopic investigation of the origin of suspended particulate matter (SPM) in the Murray–Darling river system, Australia. *Chem. Geol.*, 59: 3799–3815.

Douglas G., Palmer M. & Caitcheon G. 2003. The provenance of sediments in Moreton Bay, Australia: A synthesis of major, trace element and Sr–Nd–Pb isotopic geochemistry, modelling and landscape analysis. *Hydrobiologia,* 494: 145–52.

Fry B. & Sherr E.B. 1984. $\delta^{13}C$ measurements as indicators of carbon flow in marine and freshwater ecosystems. *Contrib. Mar. Sci.,* 27: 13–47.

Gibbs M.M. 2008. Identifying source soils in contemporary estuarine sediments: A new compound-specific isotope method. *Est. Coasts,* 31: 344–359.

Hancock G. & Caitcheon G. 2010. *Sediment sources and transport to the Logan-Albert River estuary during the January 2008 flood event. Water for a healthy country national research flagship.* (http://www.clw.csiro.au/publications/waterforahealthycountry/2010/wfhc-Logan-Albert-River-flood.pdf).

Hancock G.J., Wilkinson S.N. & Read A. 2007. *Sources of sediment and nutrients to the Gippsland Lakes assessed using catchment modelling and sediment tracers.* CSIRO Land and Water Science Report 70/07. (http://www.clw.csiro.au/publications/science/2007/sr70-07.pdf).

Hu J., Zhang H. & Peng P. 2006. Fatty acid composition of surface sediments in the subtropical Pearl River estuary and adjacent shelf, Southern China. *Est. Coast. Shelf Sci.*, 66: 346–356.

Lamb A.L., Wilson G.P. & Leng M.J. 2006. A review of coastal palaeoclimate and relative sea-level reconstructions using delta^{13}C and C/N ratios in organic material. *Earth Sci. Rev.,* 75: 29–57.

Olley J.M., Murray A.S., Mackenzie D.H. & Edwards K. 1993. Identifying sediment sources in a gullied catchment using natural and anthropogenic radioactivity. *Water Resour. Res.,* 29: 1037–1043.

Peterson B.J. & Fry B. 1987. Stable isotopes in ecosystem studies. *Ann. Rev. Ecolog. Syst.*, 18: 293–320.

Phillips D.L., & Gregg J.W. 2003. Source partitioning using stable isotopes: coping with too many sources. *Oecologia,* 136: 261–269.

Prahl F.G., Ertel J.R., Goni M.A., Sparrow M.A. & Eversmeyer B. 1994. Terrestrial organic carbon contribution to sediments on the Washington margin. *Geochim. Cosmochim. Acta,* 58: 3035–3048.

Rustomji P., Caitcheon G. & Hairsine P. 2008. Combining a spatial budget with geochemical tracers and river station data to construct a catchment sediment budget. *Water Resour. Res*., 44: WO1422, 16pp.

Seki O., Nakatsuka, T., Shibata H. & Kawamura K. 2010. A compound specific n-alkane $\delta^{13}C$ and δD approach for assessing source and delivery processes of terrestrial organic matter within a forested watershed in northern Japan. *Geochim. Cosmochim. Acta*., 74: 599–613.

Thurman E.M. 1985. *Geochemistry of natural waters*. Development in Biogeochemistry Series, Dordrecht, Kluwer. 388 pp.

Wallbrink P.J., Murray A.S. & Olley J.M. 1998. Determining sources and transit times of suspended sediment in the Murrumbidgee River, New South Wales, Australia, using fallout ^{137}Cs and ^{210}Pb. *Water Resour. Res*., 34: 879– 887.

Wallbrink P.J., Olley J.M., Murray A.S. & Olive L.J. 1996. *The contribution of subsoil to sediment yield in the Murrumbidgee River basin, New South Wales, Australia. Erosion and sediment yield.* Global and Regional Perspectives. Proceedings Exeter Symposium, pp. 347–355. IAHS Publication, vol. 236. Wallingford, UK, IAHS Press.

Walling D.E. 2005. Tracing suspended sediment sources in catchments and river systems. *Sci. Total Environ*., 344: 159–184.

Walling D.E. & Woodward J.C. 1992. *Use of radiometric fingerprints to derive information on suspended sediment sources. Erosion and Sediment Transport Monitoring Programmes in River Basins*. Proceedings Oslo Symposium, pp. 153–164. IAHS Publication, vol. 210. Wallingford, UK, IAHS Press.

Tracing Crop-Specific Sediment Sources in Agricultural Catchments with Compound-Specific Stable Isotope (CSSI) and Geochemical Markers

W.H. Blake[1,*], K.J. Ficken[2], P. Taylor[1], M.A. Russell[3] and D.E. Walling[4]

ABSTRACT

Compound specific stable isotopes (CSSI) in soils have the potential to trace suspended sediment in river channels back to source areas determined by specific crop type. As such they offer to provide support for soil resource management policies and also to inform sediment risk assessment for the protection of aquatic habitats and water resources. This manuscript summarises a study which aimed to assess the potential for CSSI tracers to provide information on sediment sources in mixed land use agricultural catchments in the United Kingdom (UK). Source discrimination was undertaken in two ways using: (i) CSSI properties and (ii) more conventional geochemical markers. The properties of the source areas were compared with those of suspended sediment collected during a major storm event using established unmixing models. Results from both methods suggested that grassland was an important source of sediment (ca. 0.13 ± 0.02 t/ha) but the geochemical fingerprinting method appeared to be slightly biased to cultivated sources owing to the influence of prior ploughing on the geochemical signals in soil of temporary pasture fields (ley pasture) that were in rotation. With prior knowledge of ley and permanent pasture management practice, analysis of the discrepancies between the two sets of tracers highlights the importance of damaged permanent pasture as an erosion hotspot. Compound specific stable isotopes offer a powerful means of addressing the impacts of land management on soil erosion and downstream sediment problems, but further research is required into the influence of crop rotations on CSSI signature development and implications for downstream interpretation.

Key words: ***compound specific stable isotopes, geochemical markers, sediment sources, erosion hotspots.***

INTRODUCTION

Compound specific stable isotope (CSSI) sediment tracing technology offers a new and exciting tool to inform soil and sediment management (Gibbs, 2008, Blake *et al.*, 2012, Hancock and Revill, 2013). The basic principal of the approach is akin to more established sediment fingerprinting and tracing methodologies but offers an additional dimension in terms of tracing material to specific crop type as opposed to generic source end members e.g. surface or subsurface material.

Sediment tracing methods, in general, rely on the ability to characterise and discriminate material from different sediment source areas using a suite of natural tracer properties (e.g. geochemical, mineral magnetic, radiochemical or biomarkers). If this can be achieved, then the properties of material from downslope or downstream locations can be compared with the source areas and relative contributions determined, resulting from effective precipitation (PPT) events (see Figure 1; Collins and Walling, 2002), subject to some important assumptions. The most important assumption made is that sediment fingerprint properties are not transformed during transportation or storage in the river or stream system.

Gibbs (2008) first demonstrated the potential use of CSSIs to trace estuarine sediment back to specific catchment sources, in this case to demonstrate the importance of forestry practice in delivering silt to an estuary. The study showed that CSSI signatures of particle-associated plant fatty acids (FAs) under exotic pine plantation were statistically different from these of soils under native forest cover and agricultural land uses. This results from the fact that FAs readily leach from plant leaves and roots and bind strongly to surface mineral particles in the upper soil profile. Of importance to tracer applications, sediment-associated FAs carry a consistent carbon-13 isotopic signature (delta C-13, $\delta^{13}C$) that reflects the C isotope fractionation processes within the source soil plant cover. Different plant types produce the same particle-reactive compounds (e.g. FAs) but with different CSSI signatures (Chikaraishi and Naraoka, 2003). In terms of satisfying the key assumption that tracer properties are not transformed between the source and receptor landscape units, it has been suggested that the $\delta^{13}C$ signature of the organic compounds does not change or degrade over time since volatilisation, dilution, dispersion and sorption do not cause isotopic fractionation (Blessing, Jochmann and Schmidt, 2008). Evidence has, however, been presented that shorter chain lipids can be degraded preferentially by soil microorganisms (Matsumoto *et al.*, 2007) which has implications for the way CSSI signatures are used and interpreted. Perhaps of greater concern is the influence of legacy effects of prior land use where fields are in a crop rotation system. These factors all require further investigation. Regarding physical factors that might influence properties, it is generally believed that CSSI properties are not modified by transport processes or fluvial sorting. In the case of geochemical tracer properties, the effect of fluvial sorting and particle size effects

1 School of Geography, Earth and Environmental Sciences, Plymouth University, Plymouth, Devon, PL4 8AA, United Kingdom
2 Department of Geography, College of Science, Swansea University, Singleton Park, Swansea, SA2 8PP, United Kingdom
3 Environment Agency, Exminster House, Exminster, Devon, EX6 8AS, United Kingdom
4 College of Life and Environmental Sciences, University of Exeter, Exeter, EX4 4RJ, United Kingdom
* E-mail address of corresponding author: william.blake@plymouth.ac.uk

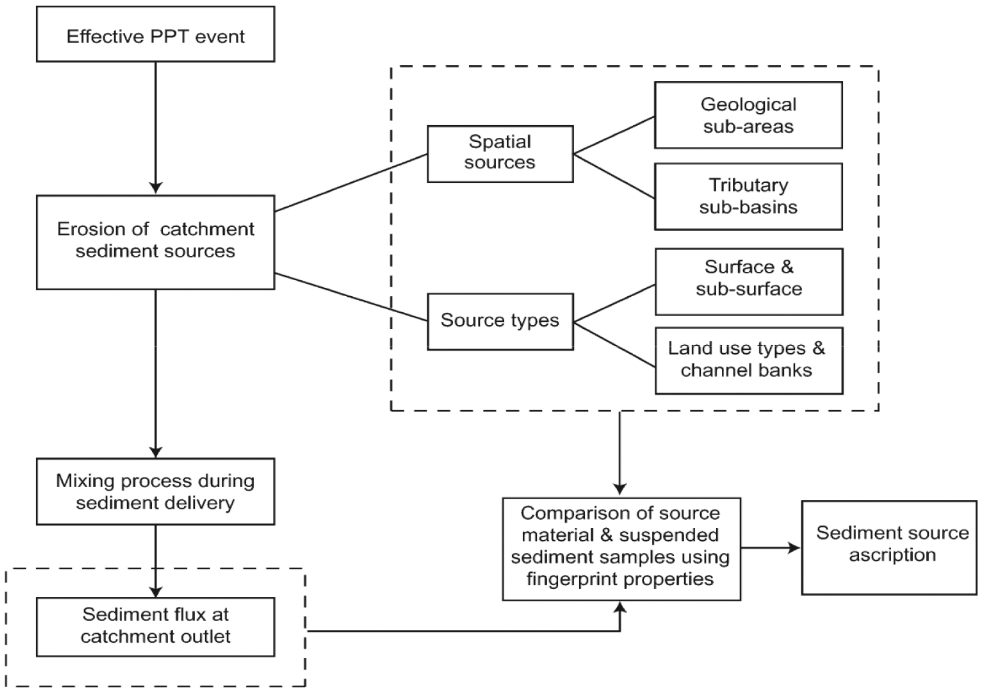

FIGURE 1. Schematic overview of the sediment fingerprinting concept (reproduced from Collins and Walling, 2002).

on sediment properties has been shown to influence fingerprint properties in some systems leading to particle size correction factors begin applied in many studies (Walling, 2005). As CSSI properties are not particle size specific, no such correction is required.

Against this background, this study aimed to explore the potential for using CSSIs as tracers in a United Kingdom (UK) agricultural context and further, to compare the results from CSSI tracing with those from more conventional geochemical tracers.

MATERIALS AND METHODS

The study was undertaken in a small (145 ha) catchment in southwest UK (Figure 2). The land cover at the time of sampling was dominated by pasture (86 ha) cover.

Land use at the time of study comprised a mix of 69 ha ley pasture (temporary grassland as part of a crop rotation system) and 17 ha permanent pasture (i.e. not previously cultivated). Cultivated land was maize (24 ha) and winter wheat (16 ha). The upper catchment was wooded (ca 5 ha) and riparian zones in places were fringed by deciduous trees. In order to characterise the material from the source areas, spatially-integrated samples of soil were collected from fields with maize, winter wheat, permanent pasture and ley pasture cover. For each sample, random transects were laid out in targeted fields and soil material collected at 1 m intervals was bulked into one bag. Integrated samples from deciduous woodland and channel banks were also taken. In total, 62 samples of soil and seven channel bank samples were collected. Samples were dried at <45°C prior to being gently disaggregated and sieved. The <63 micron fractions of the source materials were analysed for CSSI properties and also a suite of major and minor element geochemical properties. Suspended sediment samples were collected during a significant rainstorm event (24.4 mm over 5 hours). Bulk samples of sediment and water were collected at regular intervals during the storm hydrograph and returned to the lab for centrifugation, drying, sieving and CSSI and geochemical analysis as above. Flow and suspended sediment concentration were monitored at the catchment outlet at 15-minute intervals.

To comply with Gibbs (2008), bulk $\delta^{13}C$ ratios for specific extracted fatty acids were determined. Geochemical properties were determined by standard ICP-MS analysis and particle size properties by laser granulometry. Full analytical details are provided in Blake *et al.* (2012).

CSSI and geochemical data were used to discriminate slightly different, but related, source end members in accord with the theoretical basis of each approach (Figure 3). Whereas the CSSI end members are in theory defined exclusively by crop and plant cover, the geochemical signatures are controlled by vertical and horizontal variability in the geochemical properties of the substrate. In the case of cultivated *versus* uncultivated land, this is largely due to mixing of the weathering profile. In this specific catchment, the main area of woodland in the upper catchment had a different geological substrate. When comparing results from the different methods, it is essential to have a good theoretical underpinning of signature development.

Geochemical data were processed and unmixed following standard approaches based on Collins and Walling, (2002). CSSI data were processed and unmixed following the approaches outline by Gibbs (2008) using the Isosource model (Phillips and Gregg, 2003).

RESULTS AND DISCUSSION

Source discrimination

Bulk $\delta^{13}C$ of soil from under the different crop types showed some discrimination between the identified end members (Figure 4), but the extent of discrimination expected from the literature was not present. This is especially notable when comparing maize, a C4 plant, to winter wheat, a C3 plant, since the maize values would be expected to be less negative. This implies that the bulk carbon in the soil comprises material derived from previous crop rotations and also potentially weeds and cover crops.

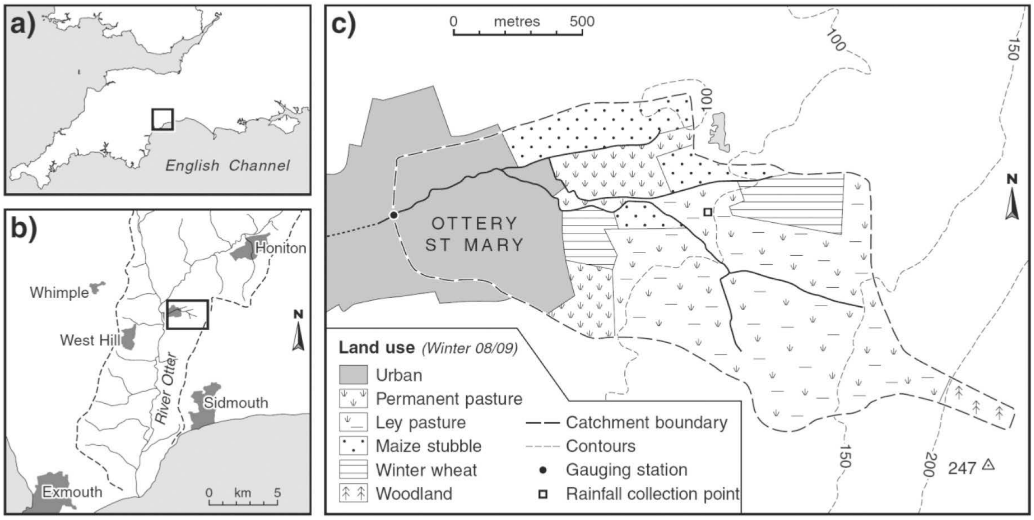

FIGURE 2. Study catchment location (a and b) and land cover (c) at the time of sampling (reproduced from Blake *et al.*, 2012).

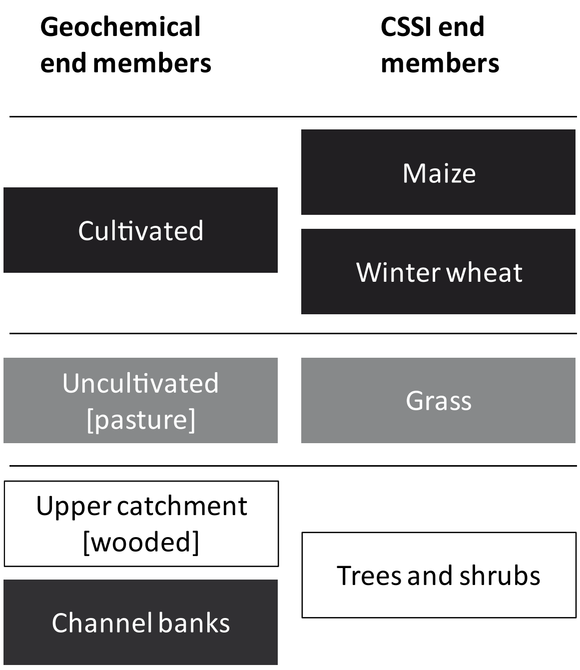

FIGURE 3. Relationship between the different source end members as defined by geochemical and CSSI approaches.

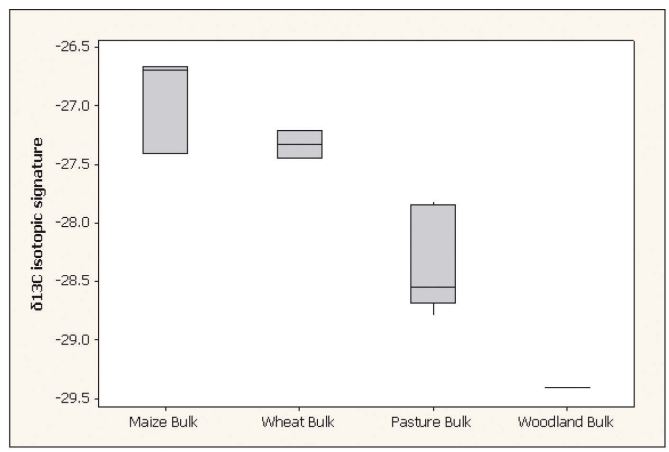

FIGURE 4. Range of $\delta^{13}C$ signatures for bulk C from different source end members.

The CSSI signatures of the soil, however, showed more promising discrimination between the land cover types in shorter chain length (C-16 and C-18) fatty acids. The range of $\delta^{13}C$ values for soil beneath maize was notably different than that for winter wheat in the C-18 length fatty acids (Figure 5). While winter wheat and grassland showed overlap, grassland had a greater range of values with many sites showing more negative $\delta^{13}C$ values than winter wheat. Woodland samples were limited in number, but showed better discrimination in C-16 chain length data (not shown). The longer chain fatty acids did not show the same degree of discrimination as seen in C-16 and C-18 groups. This was slightly surprising as the same signature was expected in all compounds. If microbial action (Matsumoto *et al.*, 2007) is a factor it could be that legacy short-chained fatty acids from previous crops were broken down, whereas the longer-chain fatty acids from previous crops remained in the soil, in line with the bulk $\delta^{13}C$ data. Alternatively, the observations could relate to other ecological activities in the soil specific to the cultivation practice. Further work is required to explore the internal discrepancies between the CSSI signatures of different fatty acids, but in the context of the study aims, the discrimination seen allowed the study to move to the next stage, i.e. apportionment of sediment.

The geochemical properties offered discrimination based on major and trace element components of the mineral fraction (see Blake *et al.*, 2012 for full details). Generally, cultivated soils were discriminated by lower acid-soluble concentrations of metals linked to a greater proportion of primary minerals in the cultivated profile. Channel banks were notably labelled by palladium (Pd) derived from road runoff which is channelled into the stream network. Importantly, the basis for discrimination between the cultivated and uncultivated soil indicates that ley pasture signatures are likely to reflect past cultivation, i.e. the approach will potentially overestimate contributions from cultivated land at the time of sampling.

Source apportionment

Comparison of the CSSI properties of suspended sediment samples (Figure 6) with source materials enabled the temporal dynamics

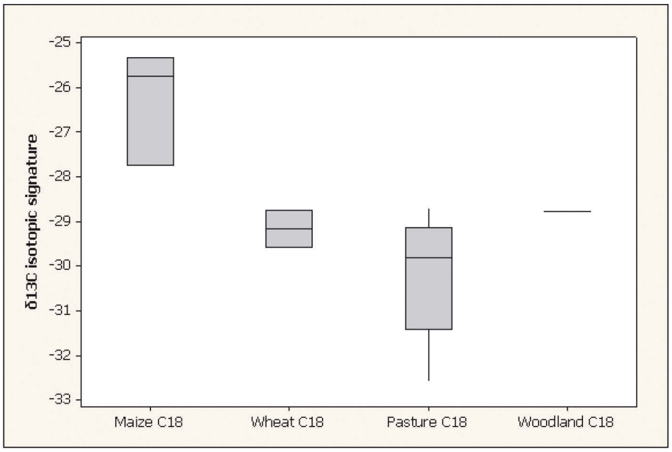

FIGURE 5. Range of CSSI signatures ($\delta^{13}C$) for C-18 chain length fatty acids from different source end members.

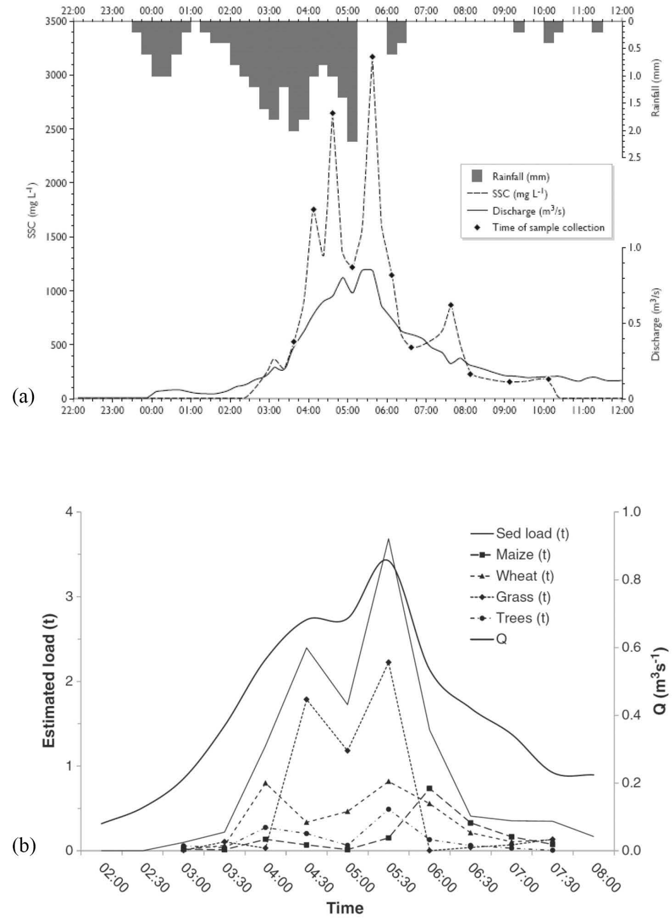

FIGURE 6. (a) Hydrograph and suspended sediment sampling points for the studied storm event and (b) sediment load apportioned to specific crops according to CSSI data (reproduced from Blake *et al.,* 2012).

of sediment delivery from different crop types to be explored. Soil from grassland sources dominated the sediment load with a near synchronous pattern in discharge and sediment. This implies a rapid response from the grassland which is corroborated by a parallel study involving rainfall simulation experiments in the catchment (Hogan, Deeks and Read, 2009). That work showed that pasture slopes were highly compacted with a rapid response to rainfall. Maize and wheat inputs were fairly continuous, in line with compaction due to farm machinery (on tractor tram lines), with a late peak in the maize input suggesting saturation driven run-off from the main field areas. This is again corroborated by rainfall simulation data (Hogan, Deaks and Read, 2009).

In terms of total sediment yield during the event, results from the CSSI data (Table 1) indicated that pasture was a dominant source of suspended sediment in the catchment. This important contribution to total sediment load was a function of the large area of grassland (65 percent) within the catchment at the time of sampling, within which compaction and poaching of the soil surface by livestock are likely to have enhanced the potential for sediment mobilization (Russell, Walling and Hodgkinson, 2001; Bilotta, Brazier and Haygarth, 2007).

The areal yield from this land cover (0.13 ± 0.02 t/ha) appeared quite high in the context of literature values for gross pasture erosion rates in southwest UK, e.g. 0.1. – 0.4 $t \cdot ha^{-1} \cdot yr$ (Walling *et al*., 2006), which implies high connectivity in this system. Grassland erosion was lower than that of winter wheat at 0.44 ± 0.15 t/ha, but similar to that from maize (0.14 ± 0.02 t/ha). According to CSSI data alone, winter wheat displayed the greatest erosion rate and sediment delivery at the time of the study. Consideration of the geochemical data results, however, offers additional insights into the sediment delivery patterns during the storm event.

Geochemistry-based source specific sediment yield data (Table 2) did not fully cohere with CSSI data. The sum of maize and winter wheat yields from the CSSI results (10.4 ± 3.0 t), is ca. 3 t lower than the yield from cultivated areas indicated by the geochemical data. Likewise, the CSSI derived grassland yield was 11.0 ± 1.4 t, some 2 t greater than the geochemistry derived yield. This can be linked to the influence of prior cultivation of the ley pasture, i.e. the geochemistry-based approach identified some ley pasture as cultivated. Given that the area of permanent pasture was known, i.e. 17 ha of the total 86 ha of grassland, it can be inferred that ley pasture contributed, as a minimum, the 3 t discrepancy noted above. If this is the case, then the remainder of the grassland derived material (a maximum of ca 9 t) must have been derived from the 17 ha of permanent pasture, which implies a sediment yield of up to 0.5 t/ha. This has important implications for grassland management in the catchment. Eighty five percent of the sites examined by Hogan, Deaks and Read (2009) had "very slow" (<1 mm/h) to "slow" (<5 mm/h) infiltration rates, based

TABLE 1. Storm sediment yield from different crops types based on CSSI tracers

	ha	t	t/ha	kg/m^2
Maize stubble	24	3.4 ± 0.5	0.14 ± 0.02	0.01
Winter wheat	16	7.0 ± 2.5	0.44 ± 0.15	0.04
Grassland	85	11.0 ± 1.4	0.13 ± 0.02	0.01
Trees/shrubs	5	2.7 ± 1.5		

TABLE 2. Sediment yield from different crops types based on geochemical tracers

	ha	t	t/ha	kg/m^2
Cultivated	40	13.3 ± 0.6	0.34 ± 0.02	0.03
Pasture	85	8.9 ± 1.2	0.10 ± 0.01	0.01
Woodland	5	0.5 ± 0.2	0.10 ± 0.03	0.01
Channel banks	n/a	1.3 ± 0.8		

on the categories given by Landon (1991), leading to high surface run-off rates. The combined results of the two tracing approaches suggest that permanent pasture fields where the surface is damaged can act as an erosion hotspot.

CONCLUSIONS

The $\delta^{13}C$ of particle-reactive fatty acids extracted from soil enabled sediment in streams to be linked back to fields under specific crop cover. Furthermore, the combined CSSI and geochemical tracing approach used in this study provided unique insights into sediment source and delivery dynamics that could not have been derived from each tracer alone. Damaged permanent pasture was identified as an erosion hotspot with sediment yields equivalent to those from a juvenile winter wheat crop. The evidence presented by this and other studies within FAO/IAEA Coordinated Research Project D1.20.11 suggests that CSSIs in combination with fallout radionuclide and other tracers have great potential to identify erosion hotspots in agricultural river basins. This study has also raised some important methodological considerations and research questions about the development and application of CSSI biomarkers. In particular, the effect of crop rotation and legacy signals from prior cultivation practices needs attention. Linked to this, future work needs to explore how soil microbial action and the biodegradation of fatty acids might affect soil and sediment signatures.

ACKNOWLEDGEMENTS

The support of the FAO/IAEA Coordinated Research Project D1.20.11, and associated colleagues under Research Agreement IAEA contract UK/15538 is gratefully acknowledged. Field sampling was supported within the framework of a Masters research programme in Sustainable Environmental Management (PT) at Plymouth University. Landowners are gratefully acknowledged for access for sampling. Ben Meredith and Tim Shipton assisted with project planning and river gauging. Ellen Petticrew and Nick Reiffarth are acknowledged for discussion of the data and insights. Elsevier kindly permitted reproduction of Figures 1, 2 and 6 from published work.

REFERENCES

Bilotta, G.S., Brazier, R.E. & Haygarth, P.M. 2007. The impacts of grazing animals on the quality of soils, vegetation, and surface waters in intensively managed grasslands. *Adv. Agron.,* 94: 237–280.

Blake, W.H., Ficken, K.J., Taylor, P., Russell, M.A. & Walling, D.E. 2012.Tracing crop-specific sediment sources in agricultural **catchments.** *Geomorphology***,** 139–140: 322–329.

Blessing, M., Jochmann, M.A. &Schmidt, T.C. 2008. Pitfalls in CSSI analysis of environmental samples. *Analyt. Bioanalyt. Chem*., 390: 591–603.

Collins, A.L. & Walling, D.E. 2002. Selecting fingerprint properties for discriminating potential suspended sediment sources in river basins. *J. Hydrol.,* 261: 218– 244.

Chikaraishi, Y. & Naraoka, H. 2003. Compound specific $\delta^{13}C$ analyses of n-alkanes extracted from terrestrial and aquatic plants. *Phytochem*. 63: 361–371.

Gibbs, M. 2008. Identifying source soils in contemporary estuarine sediments: A new compound-specific isotope method. *Estuaries Coast,* 31: 344–359.

Hancock, G.J. & Revill, A.T. 2013. Erosion source discrimination in a rural Australian catchment using compound-specific isotope analysis (CSIA). *Hydrol. Process.,* DOI: 10.1002/hyp.9466.

Hogan, D.V., Deeks, L.K. & Read, R.D. 2009. *A study of soil structure infiltration and runoff in the Furze Brook catchment.* Report to the UK Environment Agency.

Landon, J.R. ed. 1991. *Booker tropical soil manual*. *A handbook for soil survey and agricultural land evaluation in the tropics and subtropics,* Chapter 6, p 69. Booker/Tate Ltd., Harlow, Essex, UK. 474 pp.

Matsumoto, K., Kawamura, K., Uchida, M. & Shibata, Y. 2007. Radiocarbon content and stable carbon isotopic ratios of individual fatty acids in subsurface soil: Implication for selective microbial degradation and modification of soil organic matter. *Geochem. J*., 41: 493–500.

Phillips, D.L. & Gregg, J.W., 2003. Source partitioning using stable isotopes: coping with too many sources. *Oecologia* 136: 261–269.

Russell, M.A., Walling, D.E. & Hodgkinson R.A. 2001. Suspended sediment sources in two small lowland agricultural catchments in the UK. *J. Hydrol.,* 252: 1–24.

Walling, D.E., 2005. Tracing suspended sediment sources in catchments and river systems. *Sci. Total Environ.,* 344: 159–184.

Walling, D.E., Collins, A.L., Jones, P.A., Leeks, G.J.L. & Old, G. 2006. Establishing fine-grained sediment budgets for the Pang and Lambourn LOCAR catchments, UK. *J. Hydrol.,* 330: 126–141.

Compound-Specific Stable Isotope Techniques for Improving Soil Conservation Strategies: An Overview of the Lessons Learned from an FAO/IAEA Coordinated Research Project

G. Dercon[1,*], M. Gibbs[2], F. Rasche[3], C. Brandt[3], G. Cadisch[3], P. Boeckx[4], S. Bodé[4], G. Hancock[5], W. Blake[6], D.E. Walling[7], L. Mabit[1], M. Benmansour[8], N. Lam[9], D. Lobb[10], P. Schuller[11], V. Golosov[12], X. Zhang[13], Y. Li[14], W. Froehlich[15], A. Asfary[16] and M. L. Nguyen[1]

ABSTRACT

This paper summarises key findings from a five-year (2009–2013) coordinated research project (CRP) on "Integrated Isotopic Approaches for Area-wide Precision Conservation to Control the Impacts of Agricultural Practices on Land Degradation and Soil Erosion", organized and funded by the International Atomic Energy Agency through the Joint FAO/IAEA Division of Nuclear Techniques in Food and Agriculture. The project brought together fifteen participants, from Australia, Belgium, Canada, Chile, China, Germany, Morocco, New Zealand, Poland, the Russian Federation, the Syrian Arab Republic, the United Kingdom and Vietnam. The project involved the use of isotopic and nuclear techniques to assess soil erosion and develop soil conservation strategies at a landscape level. The overall objective of this CRP was to develop integrated isotopic approaches using not only fallout radionuclides (FRN) but also compound-specific stable isotope (CSSI) techniques to identify critical areas of land degradation in agricultural catchments so that effective soil conservation measures can be implemented. Compound-specific stable isotope (CSSI) techniques are based on the measurement of carbon-13 (^{13}C) natural abundance signatures of specific organic compounds (e.g. fatty acids of plant and animal origins) in the soil profile. A harmonized protocol for the application of CSSI techniques measuring ^{13}C of fatty acids extracted from soils was developed to identify critical sediment source areas and erosion hotspots at the catchment scale in a range of environments and land-use systems. The results obtained show that FRN and CSSI based techniques are complementary as fingerprints and tracers of sediment redistribution within agricultural catchments. The CCSI technique provides information on sources while FRN techniques can provide information on the extent of soil losses so that effective soil conservation measures can be targeted to criticallly degraded areas in agricultural landscape.

Key Words: *compound-specific stable isotopes, fallout radionuclides, soil erosion, precision soil conservation, agricultural landscapes.*

INTRODUCTION

Current concerns to enhance food security for an ever-growing population and to arrest widespread land degradation have highlighted the important influence of agricultural land use and management on soil erosion losses and related impacts on farmers' environments. New technologies will need to be developed and applied to better understand and manage natural and agricultural resources in agroecosystems to meet the dual goal of enhancing agricultural productivity and environmental sustainability.

Precision soil conservation is a rapidly developing key science integrating geospatial techniques, models and other tools to better understand the spatial variability of erosion across a landscape by connecting farms to natural surrounding areas and identifing erosion hot spots within agricultural lands. This integrated approach allows informed decisions on the management of critically-degraded areas and assists land managers to target soil conservation measures and appropriate land-uses to these hot spots. This will ultimately result in

1 Joint FAO/IAEA Division of Nuclear Techniques in Food and Agriculture, International Atomic Energy Agency, Vienna, Austria
2 National Institute of Water and Atmospheric Research, Hamilton, New Zealand
3 Institute of Plant Production and Agroecology in the Tropics and Subtropics, University of Hohenheim, Stuttgart, Germany
4 Faculty of Bioscience Engineering, Ghent University, Ghent, Belgium
5 Division of Land and Water, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Canberra, Australia
6 School of Geography, Earth and Environmental Sciences, Plymouth University, Plymouth, Devon, UK
7 College of Life and Environmental Sciences, University of Exeter, Exeter, Devon, UK
8 Centre National de l'Energie, des Sciences et des Techniques Nucléaires, Rabat, Morocco;
9 Hanoi University of Agriculture, Hanoi, Vietnam
10 Faculty of Agricultural and Food Sciences, University of Manitoba, Winnipeg, Canada
11 Instituto de Ciencias Químicas, Facultad de Ciencias, Universidad Austral de Chile, Valdivia, Chile
12 Faculty of Geography, Lomonosov Moscow State University, Moscow, Russia
13 Institute of Mountain Hazards and Environment, Chengdu, Sichuan, P. R. China
14 Institute of Environment and Sustainable Development in Agriculture, Beijing, P. R. China
15 Institute of Geography and Spatial Organization, Polish Academy of Sciences, Poland
16 Department of Soil Science and Reclamation, Faculty of Agriculture, University of Aleppo, Aleppo, Syria
* E-mail address of corresponding author: g.dercon@iaea.org

L.K. Heng, K. Sakadevan, G. Dercon and M.L. Nguyen (eds), Proceedings — International Symposium on Managing Soils for Food Security and Climate Change Adaptation and Mitigation. Food and Agriculture Organization of the United Nations, Rome, 2014: 149–151

improved efficiency of resource use, economic returns and environmental sustainability.

OVERVIEW ON COMPOUND-SPECIFIC STABLE ISOTOPES TECHNIQUES FOR PRECISION SOIL CONSERVATION

Through the IAEA funded coordinated research project (CRP) entitled "Integrated Isotopic Approaches for Area-wide Precision Conservation to Control the Impacts of Agricultural Practices on Land Degradation and Soil Erosion (2009–2013)", an approach based on compound-specific stable isotope (CSSI) techniques was identified for further development to support the implementation of precision conservation. These techniques are based on the measurement of carbon-13 (^{13}C) natural abundance signatures of specific organic compounds (e.g. fatty acids of plant and animal origins) in the soil profile. By linking fingerprints of land use to the sediment in deposition zones, CSSI techniques have been shown to be useful tools for identifying the source of eroded soil or transported sediment and thereby identifying areas sensitive to land degradation/erosion (Gibbs, 2010; Blake *et al*., 2012).

Significant progress in the development of these techniques was made in this CRP, as indicated below:

- A harmonized protocol was developed for the application of CSSI techniques using ^{13}C signatures in soil fatty acids to identify critical sediment source areas and erosion hotspots at the catchment scale in a range of environments and land-use systems (Gibbs, 2010). This protocol was tested successfully in Australia, Austria, Belgium, Canada, Ethiopia, Germany, New Zealand, United Kingdom and Vietnam, and was further tested by seven other countries across the world in 2013
- A summary was produced of the types, advantages and limitations of organic compounds that can be used as fingerprints to identify sediment sources and critical areas of soil loss (Table 1).
- Bulk ^{13}C signatures in soils (deposited sediment) can provide general information about sources of soil losses, but ^{13}C signatures in soil fatty acids offer important opportunities for obtaining more precise and detailed information. Bulk nitrogen-15 (^{15}N) might also help with broad-brush identification.
- With regards to CSSI techniques, the analysis requires specific intruments and skilled operators (linking gas-chromatography via an on-line combustion interface to isotope ratio mass spectrometry or GC-c-IRMS), and therefore an analytical service provider is needed. While current costs of analysis are high, these are expected to drop dramatically in the coming years, reducing to a as little as US$65 per sample.

OVERVIEW ON THE INTEGRATION OF COMPOUND-SPECIFIC STABLE ISOTOPES AND FALLOUT RADIONUCLIDES TECHNIQUES FOR PRECISION SOIL CONSERVATION

The CRP team also assessed the advantages and the way forward for the integration of CSSI with fallout radionuclide (FRN) based techniques. As FRNs have been proven to be powerful tools for assessing landscape-wide soil redistribution and identifying erosion processes, their integration with CSSI analysis will open new opportunities for improving area-wide soil conservation strategies (Gibbs, 2010; Dercon *et al*., 2012). For instance, Gibbs (2010) reported that the use of beryllium-7 (^{7}Be), an FRN with a short half-life of 53 d, allows identification of recent sediment deposits. Information obtained can then be used to collect sediment samples for CSSI analysis so that hotspots of recent land degradation (sources and intensity of soil loss) in the New Zealand study site can be identified. In addition, the linking of FRNs such as caesium-137 (^{137}Cs) and lead-210 (^{210}Pb) (half-lives of 30 and 22 years, respectively) with CSSI analysis showed how past land degradation and its link with land-use history over the last hundred years can be reconstructed (Gibbs, 2010). Combined use of FRNs and CSSI has also been used successfully in Australia to identify hot spots and assess changes in soil erosion–deposition (sediment dynamics) across a landscape (e.g. relative importance of channel bank erosion and sources).

TABLE 1. Summary of suitable specific organic compounds to identify and apportion areas sensitive to erosion through CSSI analysis.

Type	Origin	Applications	Advantages	Limitations
Fatty acids (^{13}C)	Root exudates; plant materials; animals	Land-use soil source identification to the root depth	Isotopic signature is conservative. Fatty acids are polar, move deep into the soil with water and are tightly bound to clays.	Concentrations may be low in older or sandy soils requiring larger sample analysis. Mixed land-use history (frequent crop rotations) may be difficult to resolve.
Alkanes (^{13}C)	Leaf waxes	Surface soil discrimination by land-use. Top layer only.	Isotopic signature is conservative. Non-polar and hence do not move by water. Waxes are adsorbed onto surface soil layers only.	Surface layer will be eroded first so signature rapidly removed from source. Need for different GC column to separate.
Resin acids (^{13}C)	Pine trees	Identifying pine harvest as soil source.	Specific to pine trees. Specific resin acid half-life gives time since deposition.	Rapid decay of abietic acid in sunlight within a month
Lignin (^{13}C)	Plants	Terrestrial vs aquatic plant sources.	Discriminates terrigenous soil source proportions	—
Fatty acids (Deuterium)	Root exudates; plant materials; animals	Altitude in a single land use.	Adds new dimension. The D isotopic signatures may separate similar land uses from different altitudes with > 500 m differences (rainfall influence).	Extra cost for analysis

Overall, the results obtained from the CRP demonstrated that the combination of FRNs and CSSI provides integrated information on erosion processes, spatial distribution of eroded soils and erosion sources. The CSSI-based techniques provide information on sources and, in the context of monitoring programmes, can provide quantitative information. They are complementary to the FRN techniques as fingerprints and tracers of soil redistribution within a landscape. Thus the CCSI-based techniques provide an additional dimension to generic "soil redistribution" information that is more relevant to land-use management decision-making.

CHALLENGES IN THE USE OF COMPOUND-SPECIFIC STABLE ISOTOPIC TECHNIQUES

The need for uncertainty analysis and error propagation when combining methodologies has been considered in this CRP. Sample numbers are a key consideration and should be scaled to the size and complexity of the study area and the number of land uses. Sufficient samples (minimum five per source), are required for a robust estimate of appropriate descriptive statistics. The nature of uncertainty is likely to be different for different approaches and this needs to be considered on a case-by-case basis. In this context, a detailed knowledge of land-use history within the catchment can be useful in some situations.

For those countries without easy access to CSSI analysis, intermediate approaches should be identified. In this case, bulk isotopic signatures can be an intermediate approach for identifying hot-spots. Bulk ^{13}C signatures can provide general information about soil sources (e.g. agricultural soils are different from karst soils; C_3 versus C_4 plants), but they will not provide the more precise and detailed information provided by CSSIs. Bulk ^{15}N might also help with broad-brush identification over short time-scales (d). However, when using ^{15}N, account should be taken of the N dynamics in the agroecosystem, and advantage taken of the many factors influencing ^{15}N signatures from different land uses. Bulk ^{15}N should only be used as a tracer in flowing water in wet systems, where the ^{15}N signature may not have changed since it entered in the geochemical cycle.

CONCLUSIONS

CSSI and its integration with FRNs show potential for developing soil conservation strategies for agricultural landscapes. Combined tracer approaches will help overcome the complexities for assessing land degradation and soil distribution at a landscape level. Interdisciplinary approaches including soil science, geomorphology, hydrology and biogeochemistry potentially assist in identifying key processes of land degradation and soil distribution. Information obtained can then be used to identify the most appropriate tools and sampling strategies.

REFERENCES

Blake, W.H., Ficken, K., Taylor, P., Russell, M. & Walling, D.E. 2012. Tracing crop-specific sediment sources in agricultural catchments. *Geomorphology*, 139–140: 322–329.

Dercon, G., Mabit, L. Hancok, G., Nguyen, M.L., Dornhofer, P., Bacchi, O.O.S., Benmansour, M., Bernard, C., Froehlich, W., Golosov, V.N., Haciyakupoglu, S., Hai, P.S., Klik, A., Li, Y., Lobb, D.A., Onda, Y., Popa, N., Rafiq, M., Ritchie, J.C., Schuller, P., Shakhashiro, A., Wallbrink, P., Walling, D.E., Zapata, F. & Zhang, X. 2012. Fallout radionuclide based techniques for assessing the impact of soil conservation measures on erosion control and soil quality: An overview of the main lessons learnt under an FAO/IAEA Coordinated Research Project. *J. Environ. Radioact.,* 107: 78–85.

Gibbs, M. 2010. *Protocols on the use of compound-specific stable isotopes to identify and apportion soil sources from land use*. NIWA Report HAM 2010-129 to Joint FAO/IAEA Division of Nuclear Techniques in Food and Agriculture. 97pp.