Process interaction and sediment delivery in the Pleiser Hügelland, Germany

N. J. PRESTON[1] **& R. DIKAU**[2]

1 *Department of Physical Geography, Macquarie University, North Ryde, New South Wales 2109, Australia*
npreston@els.mq.edu.au

2 *Geographisches Institut, Universität Bonn, Meckenheimer Allee 166, D-53115 Bonn, Germany*

Abstract Sediment redistribution rates in a small agricultural catchment in the loess-covered hill country of western Germany have been modelled using ^{137}Cs. The study site comprises three parcels, two of which are cultivated and the third is pasture. The pattern of erosion and deposition is broadly controlled by water erosion, but the effect of tillage translocation is important. Tillage appears to enhance rates of water erosion within parcels, but it retards sediment export from the parcel, emphasizing the importance of land-use boundaries.

Key words configuration; ^{137}Cs; Germany; land-use; process interaction; sediment delivery; tillage; water erosion

INTRODUCTION

Tillage translocation has been recognized as an important geomorphic process (e.g. Gov94ers *et al.*, 1994). However, the broader implications of tillage translocation for sediment redistribution dynamics within catchments remain relatively unexplored. Many catchment-scale models of sediment redistribution treat geomorphic processes individually, although it is likely that interaction between processes may play an important part in determining patterns of sediment redistribution.

One of the most powerful tools available for exploring soil and sediment redistribution is ^{137}Cs (e.g. Zapata, 2002). The movement of ^{137}Cs through the landscape represents an integral signal of all sediment redistributive processes that have been active over a period of decades. For the agricultural area of the Pleiser Hügelland, east of Bonn, these principally include water erosion and tillage translocation. There are important differences between these two types of process, relating to distance of transport, size selectivity of particle entrainment and deposition, extent of occurrence and frequency/magnitude spectra. Thus land-use, insofar as it determines the processes by which ^{137}Cs in the soil may potentially be eroded and transported, is an important factor in modelling 9 sediment redistribution.

In this paper we use ^{137}Cs to explore the interaction of water erosion and tillage translocation in the Auf dem Scheid catchment. We have used ^{137}Cs to reconstruct patterns of sediment movement associated with each of these two processes. In addition, the use of a ^{137}Cs-derived mass balance shows variation in sediment delivery with land-use.

STUDY AREA AND METHODS

The Pleiser Hügelland is a predominantly agricultural area to the east of Bonn (Fig. 1). Average annual rainfall is 700 mm, with highest intensities generally occurring in the early

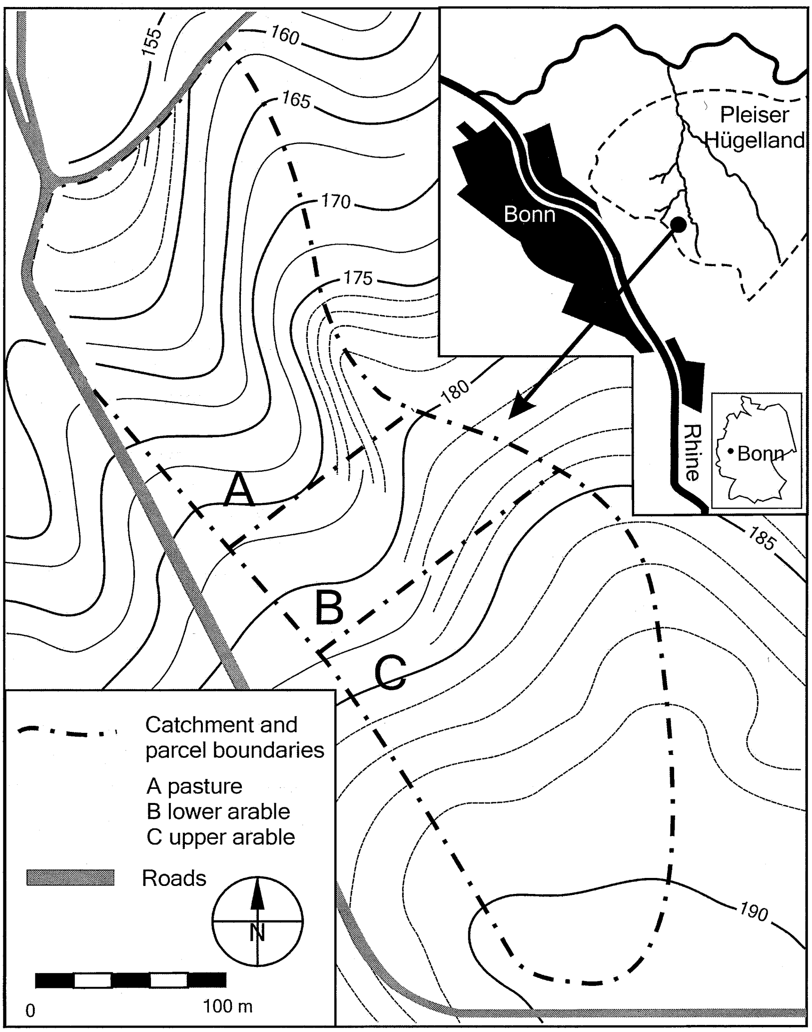

Fig. 1 Location of the study site. Note the three parcels: A, pasture; B and C, arable land.

summer months when fields are least protected by a crop cover. The study site—Auf dem Scheid—is a small (approx. 5.4 ha) zero-order basin. Soils are formed in Pleistocene loess, and most have been extensively eroded as a result of agricultural use. The lower part of the Auf dem Scheid catchment is under pasture, while the upper part is ploughed with a three-way crop rotation. The boundary between these two land-uses is marked by an abrupt break in slope, representing the accumulation of approx. 1 m of material at its greatest. A farm road dating back to at least 1940 defines the lower boundary of the catchment. This road and artificial changes to drainage lines that were undertaken in the 1950s mean that hydrological coupling between the Auf dem Scheid basin and the higher order drainage system is buffered, and under normal conditions Auf dem Scheid is a closed sediment redistribution system.

In an initial survey, ^{137}Cs was found to a depth of 0.3 m at a nearby depositional site. It was therefore considered that sampling to a depth of 0.5 m would enable recovery of all

Table 1 Values used in parameterization of sediment redistribution models.

Factor	Mass balance model	Diffusion and migration model
Proportionality factor (γ)	0.05	n/a
Relaxation mass depth (H)	3.8 kg m^{-2}	5.2 kg m^{-2}
Mass depth of the plough layer	300 kg m^{-2}	n/a
Tillage constant	720 kg m^{-1} year^{-1}	n/a
Diffusion coefficient (D)	n/a	63.5 kg m^{-4} year^{-1}
Migration rate coefficient (V)	n/a	1.13 kg m^{-2} year^{-1}
Reference inventory	5530 Bq m^{-2}	5530 Bq m^{-2}

^{137}Cs present in the soil column. Bulk samples were taken in an open cylinder with a diameter of 6.8 cm. Concentrations of ^{137}Cs per unit weight (Bq kg^{-1}) were determined by high-resolution γ-spectrometry and converted to areal activities (Bq m^{-2}).

Sediment redistribution rates were estimated using numerical models based on the behaviour of ^{137}Cs and the physical processes of sediment redistribution. A mass balance model that incorporates the tillage process was applied to the arable parcels and a migration-diffusion model to the pasture parcel. Full descriptions of these models are given by Walling & He (1999, 2001) and Walling *et al.* (2002); their parameterization is described here and summarized in Table 1.

As input the mass balance model requires:

(a) A proportional factor (γ), representing the proportion of ^{137}Cs fallout that is removed in runoff before incorporation in the plough layer. The value of γ is dependent on the timing of tillage relative to the occurrence of erosive rainfall, and has a maximum value of 1.0 in the hypothetical case of all erosive rainfall occurring immediately prior to tillage. While rainfall in the study area is relatively low, potentially erosive rainfall does typically occur when arable surfaces are at their most exposed. A low proportionality factor of 0.05 was applied to reflect the possibility that loss of ^{137}Cs in runoff may occasionally occur before tillage incorporation.

(b) A relaxation mass depth factor (H, kg m^{-2}) for the depth to which ^{137}Cs initially infiltrates when first delivered to the soil's surface. He & Walling (1997) have published empirical values of H: 3.8 kg m^{-2} for cultivated soil and 5.2 kg m^{-2} for undisturbed surfaces. The former has been adopted for use within the mass balance model for the arable zone.

(c) The mass depth of the plough layer. For Auf dem Scheid, the tillage depth is 20 cm and the average density of soils in the plough layer is 1499 kg m^{-3}. The mass depth of the plough layer is thus estimated as 299.8 kg m^{-2}, approximated to 300 kg m^{-2}.

(d) A tillage constant. This varies with type of tillage machinery and direction, timing and pattern of tillage. The tillage-induced sediment flux is generally higher for contour tillage. Lindstrom *et al.* (1992) report a value of 363 kg m^{-1} for each contour tillage operation. With two contour tillage operations per year in Auf dem Scheid, an approximate value of 720 kg m^{-1} year^{-1} has been adopted for the tillage constant.

The vertical distribution of ^{137}Cs within uncultivated soils is significantly different from that of tilled soils in which ^{137}Cs is mixed throughout the plough layer. In many cases, the vertical distribution of ^{137}Cs in the soil exhibits an exponential decline with depth. This does not remain constant, however, as ^{137}Cs slowly migrates through the soil profile. The diffusion and migration model allows for this. As input the model requires:

(a) A relaxation mass depth (*H*), representing the depth to which ^{137}Cs initially infiltrates. As with the mass balance model, a value published by He & Walling (1997) was used, i.e. 5.2 kg m^{-2} for undisturbed surfaces.

(b) A diffusion coefficient (*D*) and a migration rate coefficient (*V*) based on the depth of maximum concentration of ^{137}Cs and the depth at which concentration decreases to 1/*e* of the maximum. Detailed information on the vertical distribution of ^{137}Cs in Auf dem Scheid soils is not available, and these depths are assumed to be 5 and 11 cm respectively, based on representative profiles for silty soils published by Walling & Quine (1992). *D* is estimated to be 63.5 kg m^{-4} $year^{-1}$ and *V* to be 1.13 kg m^{-2} $year^{-1}$.

Input data describing measured ^{137}Cs inventories for sample points consists simply of the point inventories. No correction was made for particle size because of the restricted range of soil textures in the Auf dem Scheid catchment, and the empirical observation that *in situ* soils and colluvium have essentially the same textures. The mass balance model specifically applies to slope transects, and requires that input data be arranged sequentially in a downslope direction. The input file thus contains, for each point: the measured ^{137}Cs inventory (Bq m^{-2}), the length of slope segment incorporating the point and the input and output angles to and from that slope segment. Both models require input data describing the temporal distribution of ^{137}Cs fallout. This has been derived from a dataset (Cambray *et al.*, 1989) supplied with the software used to run the model (Walling & He, 2001). Chernobyl fallout in 1986 was not included in this dataset, and a value of 700 Bq m^{-2}, based on values reported by Dörr & Münnich (1987) was added to the fallout dataset.

Undisturbed sites suitable for establishment of a reference inventory were not located within the extensively agriculturally used study area. Records of fallout are not available locally, and the spatial variability of fallout receipts published for other locations is such that values cannot be assumed to apply to this study area. Schimmack *et al.* (2001) have suggested that a mean value from sampled points can give a good approximation of the total ^{137}Cs fallout receipt, if it is assumed that all ^{137}Cs remains within the study area. Given the closed nature of the Auf dem Scheid catchment over the period of ^{137}Cs deposition, this assumption is likely to be satisfied, and the sample mean of 5530 Bq m^{-2} was used as a reference inventory.

RESULTS

The distribution of ^{137}Cs activity is illustrated in Fig. 2 (n = 120, mean = 5530 Bq m^{-2}, standard deviation = 1650.35 Bq m^{-2}, standard error of the mean = 150.06 Bq m^{-2}). Variability of measured activity is high, and the 95% confidence interval for the mean is 5232–5681 Bq m^{-2}.) Of the 120 samples, 20 have activities within this range, and may have influenced the location of boundaries between erosional and depositional areas to a small extent. The pattern is based on a kriging interpolation between points with known values and is thus only indicative. Within the pasture zone there are few points with activities greatly lower than reference. More marked in this zone are the high activities at the base of slopes and especially within the thalweg. The arable zone on the other hand is characterized more by the predominance of values lower than reference. Two patterns can be recognized. In the pasture (A) and lower arable (B) parcels topography appears to exert some control over ^{137}Cs

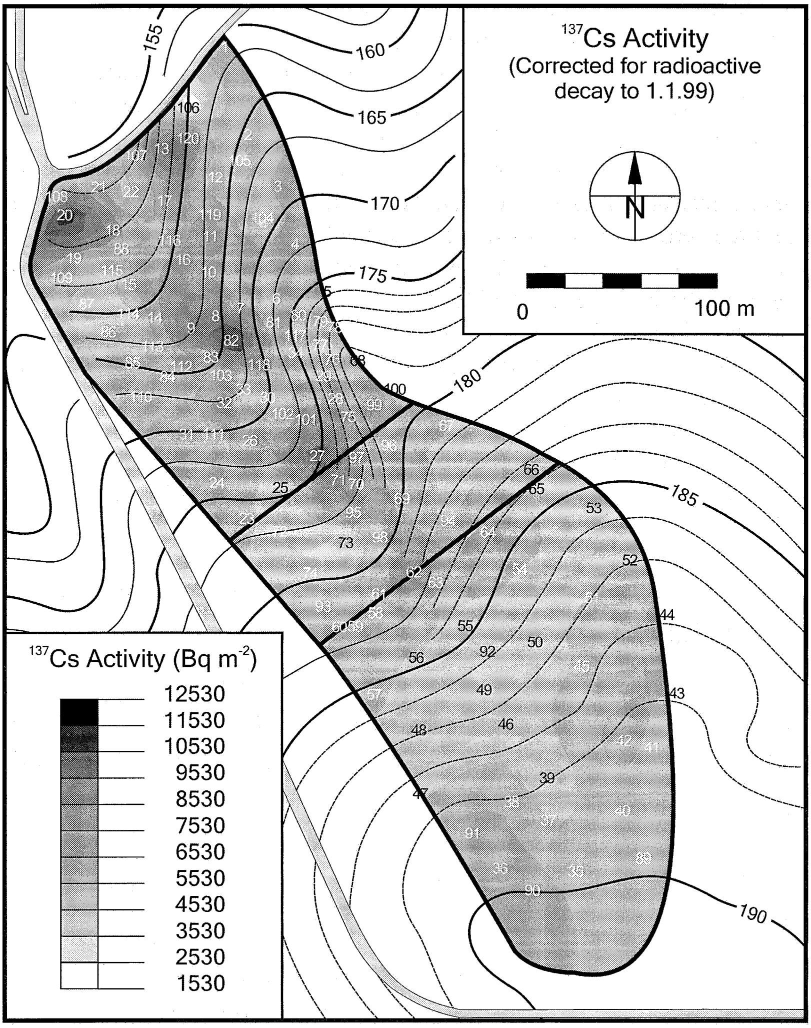

Fig. 2 Distribution of ^{137}Cs activity in Auf dem Scheid. Sample points are numbered. Intervals were chosen to indicate areas with values greater and smaller than the reference inventory of 5530 Bq m^{-2}.

activity. There is a broad relationship between topography and ^{137}Cs activity in the pasture zone, although this is by no means simple. This zone has much steeper slopes, but these do not exhibit especially low ^{137}Cs activities, and indeed in some cases show values greater than reference. Similarly, highest values in the lower arable field are in the thalweg while adjoining slopes have lower values—extremely low in the case of Points 72–74. The relationship of ^{137}Cs activity to topography appears to be reversed in the upper arable parcel (C) with values slightly higher than reference on its higher boundaries, but lower concentrations in the gently sloping middle section. A second pattern can be detected within the arable zone, reflecting its division into two fields with different cropping histories. A zone of higher activity is evident at the boundary between the two arable parcels. Again, there is a concentration of higher values at the lowest point of the boundary separating the lower arable parcel from the downslope pasture zone.

The spatial distribution of modelled net erosion and deposition rates is illustrated in Fig. 3. Clearly, there has been redistribution of sediment associated with water erosion in the pasture zone (Fig. 3(a)), but this has not been extreme, and the arable zone shows greater extremes of both erosion and deposition (Fig. 3(d)). In the arable zone the mass balance model distinguishes between rates of sediment redistribution due to both tillage translocation (Fig. 3(b)) and water erosion (Fig. 3(c)). For the greatest part of the arable zone, tillage produces relatively low rates of erosion. Deposition due to tillage, however, is restricted to a very limited area at the base of the arable zone, representing the development of a colluviation terrace at the arable field boundary. Water erosion produces a more diverse pattern of sediment redistribution and a greater range of values, especially for erosion. Different patterns can be discerned for the two parcels. The lower parcel is characterized by eroding slopes and an accumulating thalweg zone, while the upper parcel has experienced thalweg erosion and some accumulation on its almost flat flanks. The boundary between arable units represents a clear barrier to sediment transport in runoff, as best indicated by the accumulation at Points 62 and 63. This is also true of the land-use boundary dividing the catchment across its middle.

Patterns of ^{137}Cs redistribution also permit inferences to be made regarding sediment delivery. A grid of 25 m^2 cells has been overlaid on the net sediment redistribution map, and volumes of erosion and deposition within each of the three land-use units have been estimated by multiplying median values of erosion or deposition by numbers of cells within each class. Summing these provides estimates of volumes of material eroded from and accumulated within each zone and thus also of sediment delivery ratios (Table 2). The upper arable parcel is estimated to have lost to erosion an average 30.4 t $year^{-1}$ over the period 1954-1999. Only 3.7 t $year^{-1}$ was retained within this unit, which is thus a net sediment exporting zone with a high sediment delivery ratio (87.9%). The lower arable parcel is also a net sediment exporting unit, with erosional and depositional volumes of 10.9 t $year^{-1}$ and 2.5 t $year^{-1}$ respectively, and a sediment delivery ratio of 77.5%. The pasture parcel, by contrast, has accumulated material at a greater rate (6.3 t $year^{-1}$) than it has been eroded from within its boundaries (5.1 t $year^{-1}$), resulting in a negative sediment delivery ratio (–22.7%). It is difficult, however, to draw firm conclusions regarding extra-parcel sediment delivery, as there is a discrepancy between volumes of redistribution in absolute terms, apparently invalidating the closed system assumption. This assumption is considered to be justified, however, and the discrepancy may be attributable to the use of different approaches to estimate rates of sediment redistribution in each zone. Further, this sediment budget approach is highly dependent on sampling density and especially on the nature of spatial interpolation between points with known values. Given the costs and measurement time required for ^{137}Cs, 120 samples from an area of < 6 ha, with an error of mean estimation of ~5%, is considered reasonable. Inevitably, however, considerable resolution is missing, which influences the accuracy of interpolation. Comparison between the different land-use types in terms of absolute values is not valid, and it is emphasised that the rates and volumes estimated here are indicative only. Nevertheless, the patterns and results within each land-use unit, within which systematic modelling errors can be considered to be uniform, can be evaluated. These confirm that the arable units are indeed net sediment exporting units, and that the pasture zone has experienced deposition greater than can be accounted for by the sediments generated within its own boundaries.

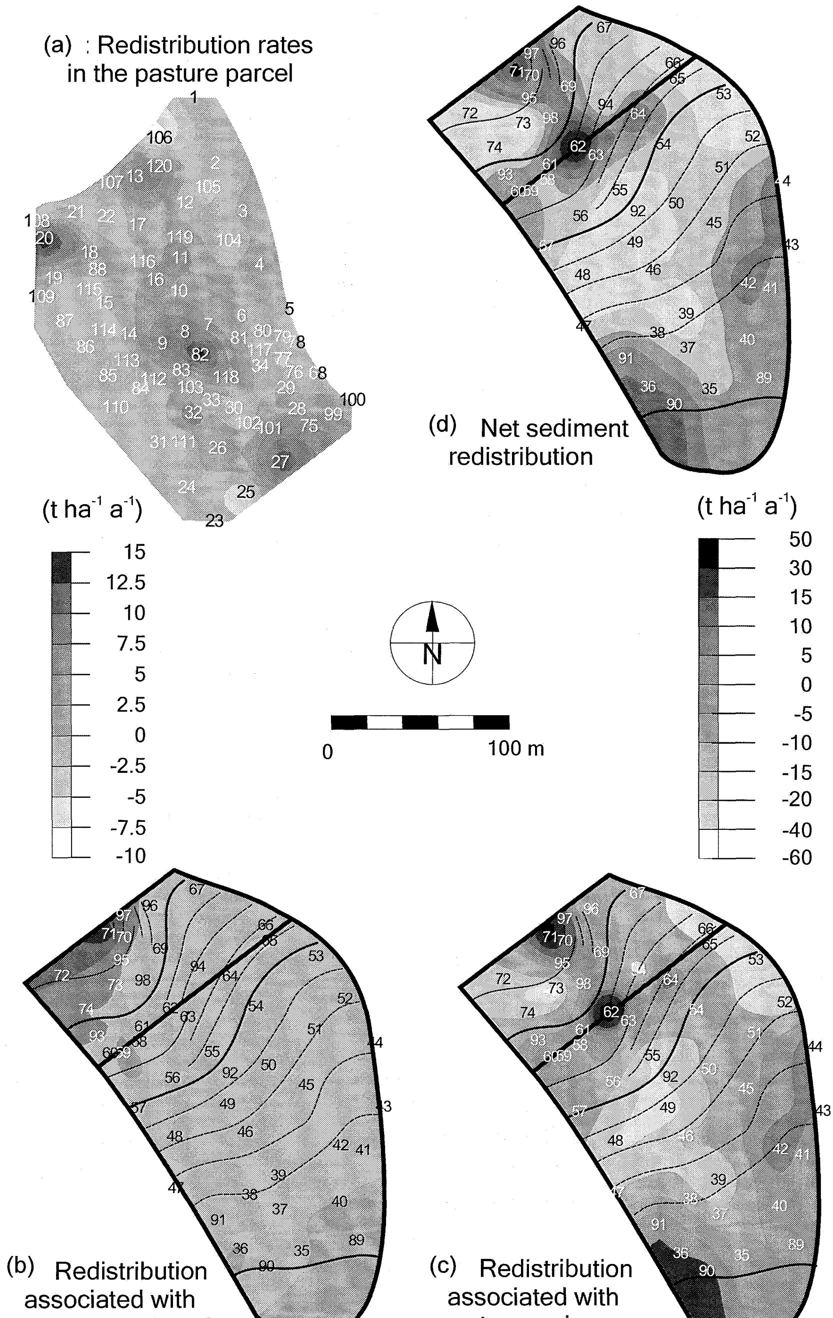

Fig. 3 Net rates of soil erosion and sediment deposition in Auf dem Scheid, modelled on the basis of ^{137}Cs activities. Sample points are numbered. Rates for the pasture zone (a) are modelled with a diffusion/migration model incorporating water erosion only, while those for the arable zone (b–d) are derived from a mass balance model parameterized for both water erosion and tillage translocation. The boundary between arable parcels is marked. Use the scale on the left for (a) and (b) and the right for (c) and (d).

Table 2 ^{137}Cs-derived sediment redistribution for the three land-use units of Auf dem Scheid.

	Class (t ha^{-1} year^{-1})	Median (t ha^{-1} year^{-1})	No. of cells	Area (ha)	Redistribution (t year^{-1})	Net (t year^{-1})	Delivery ratio
Pasture erosion	0–10	5	*410*	*1.025*	*5.125*		
Pasture deposition	0–10	5	431	1.0775	5.3875		
	10–20	15	19	0.0475	0.7125		
	20–30	25	3	0.0075	0.1875		
			453	*1.1325*	*6.2875*	1.16	–22.68
Lower arable erosion	0–10	5	59	0.1475	0.7375		
	10–20	15	108	0.27	4.05		
	20–30	25	54	0.135	3.375		
	30–40	35	13	0.0325	1.1375		
	40–50	45	9	0.0225	1.0125		
	50–60	55	4	0.01	0.55		
			247	*0.6175*	*10.8625*		
Lower arable deposition	0–10	5	37	0.0925	0.4625		
	10–20	15	20	0.05	0.75		
	20–30	25	6	0.015	0.375		
	30–40	35	6	0.015	0.525		
	40–50	45	3	0.0075	0.3375		
			72	*0.18*	*2.45*	–8.41	77.45
Upper arable erosion	0–10	5	306	0.765	3.825		
	10–20	15	280	0.7	10.5		
	20–30	25	185	0.4625	11.5625		
	30–40	35	52	0.13	4.55		
			823	*2.0575*	*30.4375*		
Upper arable deposition	0-10	5	111	0.2775	1.3875		
	10–20	15	55	0.1375	2.0625		
	20–30	25	1	0.0025	0.0625		
	30–40	35	2	0.005	0.175		
			169	*0.4225*	*3.6875*	–26.75	87.89

DISCUSSION

These results show that, in addition to its importance as a process for generating sediment and moving it over small distances, with local morphological effects, tillage may also affect rates of water erosion, and in turn influence sediment delivery dynamics. Tillage can shift soil and sediment into or out of zones of greater or lesser susceptibility to water erosion, thus impacting on the supply of material for water erosion and fluvial transport. Whether this process interaction amplifies or dampens longer-term sediment delivery is likely to depend on the relative frequency/magnitude of each process, and on the specific configuration of tillage operation relative to topography. Modelled values suggest that water erosion is the dominant of the two processes occurring on the arable part of Auf dem Scheid (Fig. 3). Tillage appears to deliver readily erodible sediments to an area in the northwestern corner of the arable zone, where it is further reworked by water erosion, with a net effect of concentrating accumulation in the thalweg outlet to the pasture zone. This suggests that in this particular case the process interaction is acting to enhance sediment redistribution.

While the pattern of net sediment redistribution within the arable zone suggests enhanced sediment redistribution through water erosion, the interaction between water erosion and tillage translocation also has implications for the whole of Auf dem Scheid. There is fluvially-derived deposition at the hydrological outlets of both arable parcels, in addition to the tillage-derived accumulation of sediment along these edges. Clearly, not all of the mobilized material is exported from the parcel. Land-use boundaries thus appear to act as barriers to transport between parcels and to have an effect on sediment delivery. An implication of this is that the interaction between processes may enhance water erosion within the parcel, but that the effect of tillage is to retard sediment export. While this has been the effect in the Auf dem Scheid catchment, it will not necessarily be the case at other locations, or with a different suite of processes or land-uses. The extent to which this is true will partly depend on the relative predominance of each process, and especially on their frequency/magnitude spectra.

Because land-use boundaries can play an important role as barriers and buffers (see also van Oost *et al.*, 2000), it is important that they be properly incorporated within catchment-scale models of sediment redistribution. This will not be trivial, as the effect is likely to be complex and contingent on subtle variations in both land-use practice and the spatial configuration of land-uses. Characterizing the nature and location of tillage is likely to be one of the more important tasks required in the use of such models.

Acknowledgements This study was conducted at Universität Bonn, within the Sonderforschungsbereich 350 project funded by the German Research Foundation. ^{137}Cs activity was measured at the Max-Planck-Institut für Kernphysik in Heidelberg.

REFERENCES

Cambray, R. S., Playford, K. & Carpenter, R. C. (1989) Radioactive fallout in air and rain: results to the end of 1988. *UK Atomic Energy Authority Report 10155.*

Dörr, H. & Münnich, K.O. (1987) Spatial distribution of soil ^{137}Cs and ^{134}Cs in West Germany after Chernobyl. *Naturwissenschaften* **74**, 249–251.

Govers, G., Vandaele, K., Desmet, P. J. J., Poesen, J. W. A. & Bunte, K. (1994) The role of tillage in soil redistribution on hillslopes. *Europ. J. Soil Sci.* **45**, 469–478.

He, Q. & Walling, D. E. (1997) The distribution of fallout ^{137}Cs and ^{210}Pb in undisturbed and cultivated soils. *Appl. Radiation Isotopes* **48**, 677–690.

Lindstrom, M. J., Nelson, W. W. & Schumacher, T. E. (1992) Quantifying tillage erosion rates due to moldboard ploughing. *Soil Till. Res.* **24**, 243–255.

Schimmack, W., Auerswald, K. & Bunzl, K. (2001) Can $^{239+240}$Pu replace ^{137}Cs as an erosion tracer in agricultural landscapes contaminated with Chernobyl fallout? *J. Environ. Radioactiv.* **53**, 41–57.

van Oost, K., Govers, G. & Desmet, P. (2000) Evaluating the effects of changes in landscape structure on soil erosion by water and tillage. *Landscape Ecol.* **15**, 577–589.

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

Walling, D.E. & He, Q. (2001) Models for converting ^{137}Cs measurements to estimates of soil redistribution rates on cultivated and uncultivated soils, and estimating bomb-derived ^{137}Cs reference inventories (including software for model implementation). *A contribution to the IAEA Coordinated Research Programmes on Soil Erosion (D1.50.05) and Sedimentation (F3.10.01).* University of Exeter, UK.

Walling, D. E. & Quine, T. A. (1992) The use of caesium-137 measurements in soil erosion surveys. In: *Erosion and Sediment Transport Monitoring Programmes in River Basins* (ed. by J. Bogen, D. E. Walling & T. J. Day) (Proc. Oslo Symp., August 1992), 143–152. IAHS Publ. 210. IAHS Press, Wallingford, UK.

Walling, D.E., He, Q. & Appleby, P.G. (2002) Conversion models for use in soil-erosion, soil-redistribution and sedimentation investigations. In: *Handbook for the Assessment of Soil Erosion and Sedimentation using Environmental Radionuclides* (ed. by F. Zapata), 111–164. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Zapata, F. (2002) *Handbook for the Assessment of Soil Erosion and Sedimentation using Environmental Radionuclides.* Kluwer Academic Publishers, Dordrecht, The Netherlands.