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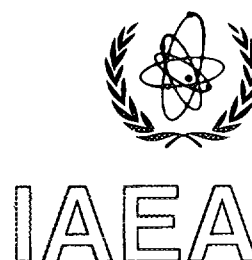
Use of ^{137}Cs in the study of soil erosion and sedimentation

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FOREWORD

Soil erosion and sediment deposition represent serious threats worldwide because of their impact on agricultural production and environmental conservation. Erosion affects the productivity of soil through loss of the nutrient-rich surface layers and the incorporation of potentially growth-limiting subsoil into the rooting zone. In many cases, erosion causes progressive decline in soil productivity, particularly so in agro-ecosystems that rely on indigenous fertility. The use of high-input technology such as large amounts of fertilizers, pesticides, and irrigation helps offset deleterious effects of erosion but has the potential to create pollution and health problems, destroy natural ecosystems, and contribute to high energy consumption and unsustainable agricultural systems.

Soil erosion causes not only on-site degradation of a precious natural resource, but also off-site problems of sediment deposition in residential areas and reservoirs, eutrophication of surface waters and pollution from various particle-adsorbed toxic agrochemicals.

Erosion and deposition are recognized to have occurred throughout the history of agriculture, and notwithstanding a half-century of research into its causes and effects, considerable uncertainty persists about extent, magnitude and actual rates, as well as on the economic and environmental consequences. When economic costs of soil loss and degradation and off-site effects are conservatively estimated into cost/benefit analyses of agriculture, it makes sound economic sense to invest in programs that are effective in the control of soil erosion.

The use of radionuclides in soil erosion/deposition research overcomes many of the problems associated with traditional approaches and is now being applied successfully in several developed countries. Among these, the ^{137}Cs technique allows the assessment of both soil loss and deposition in the same watershed from a single site visit without the need for long-term financial commitments. Caesium-137, an artificial radionuclide with a half-life of 30.2 years, is distributed across the earth's surface due to fallout from atmospheric nuclear tests and accidental releases from nuclear reactors. Strongly adsorbed by clay particles, it provides a unique tracer of soil movement.

In response to the United Nations Conference on Environment and Development convened in Rio de Janeiro in June 1992, the UN system launched a worldwide environmental programme called EARTHWATCH. The IAEA joined this initiative through a series of activities on environmental monitoring, impact assessment and environmental protection. Thus, in connection with Chapter 12, "Managing Fragile Ecosystems: Combating Desertification and Drought", of Agenda 21, two IAEA Divisions, the Joint FAO/IAEA Division of Nuclear Techniques in Food and Agriculture and the Division of Physical and Chemical Sciences, joined forces to plan, organize and implement activities on the assessment of soil erosion and sedimentation as a basis for soil conservation and environmental protection. A panel of experts on the use of isotopes in studies on soil erosion convened in November 1995 in Vienna to discuss the possibilities of exploiting radionuclide methodologies, and ^{137}Cs in particular, for the assessment of soil erosion and sedimentation in developing countries in which food security is at greatest risk. The state-of-the-art reports presented at that meeting are contained in this publication which, as the first comprehensive treatment of the subject, is expected to serve as an invaluable source of information to underpin future research on soil conservation and environmental protection. The responsible IAEA officer was F. Zapata. The assistance of D.J. Pennock and A.R.J. Eaglesham in the preparation of this publication is gratefully acknowledged.

EDITORIAL NOTE

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SUMMARY

Although current concern for the global environment focuses largely on problems of global warming and climate change, there is increasing realization that soil erosion represents a major constraint to sustainable development of agricultural production in a world characterized by a burgeoning population. Soil erosion is a natural process caused by water and wind, but the accelerative effects of human activities, most notably deforestation and poor agricultural practices, are of increasing concern. Soil erosion causes not only *on-site* effects associated with compromised soil productivity and loss of agricultural land, but there are also many *off-site* problems such as downstream sediment deposition, pollution of water courses with various adsorbed agrochemicals and eutrophication of water bodies. It is estimated that the world's arable lands are, on average, being eroded at a rate considerably in excess of that of soil formation, resulting in a net 7% depletion of soil resources each decade, with potentially serious implications for food security in many countries. Furthermore, the world's reservoirs are losing storage capacity at about 50 km³ (approximately 1%) per year, as a result of sedimentation. Considering the central importance of reservoirs for domestic and industrial water supply, for irrigation schemes, and for hydropower production, and thus for economic progress in many developing countries, this trend is of significant concern.

To date, much of the loss of soil and agricultural land through erosion has been compensated by clearing new farm land, and by use of fertilizer and improved crop varieties to increase yields on existing land. However, the scope for maintaining such compensatory measures will decline in the future.

Against this background, there is an increasing need to assemble reliable information on rates of soil loss in different areas of the world. Such information is needed to precisely assess the magnitude of the problem, to evaluate influencing factors, to validate existing and new prediction models, and to investigate relationships with crop production. This need is, however, not readily met by classical methods of measurement of erosion, which have significant limitations. They do not give unbiased measurements of soil redistribution, and, more importantly, they do not address spatial patterns of erosion and deposition within fields. There is a need for location-specific measurements within the landscape, especially in areas where other erosion data are not available and where long-term experiments have not, nor cannot be, established.

The use of radionuclides overcomes many of the problems associated with the traditional approaches and they have the potential to provide the needed data. By labelling the soil, both the extent and the source of soil loss can be determined. Several radionuclides, mainly gamma emitters have been applied as tracers in field-erosion studies (⁵⁹Fe, ⁴⁶Sc, ¹¹⁰Ag, ¹⁹⁸Au, ¹³⁴Cs, ⁵¹Cr). Another group includes the environmental radionuclides, such as ¹³⁷Cs, ²¹⁰Pb, ⁷Be, ²⁴⁰Pu, ¹⁴C, ³²Si, ²⁶Al and ³⁶Cl, which can be used to assess soil erosion and deposition patterns and related problems depending on the time scale involved.

The use of fallout radioactivity to estimate soil loss was first published in 1960, the transport of ⁹⁰Sr in runoff from "standard" erosion plots showed that tracer depletion was greatest where the most soil loss occurred. Research in the late 1960s showed that differences in distribution of fallout ¹³⁷Cs between vegetation types on a catchment in Mississippi were due to soil loss from the landscape. In the mid-1970s, the combination of data from a variety of soil and erosion conditions, with fallout radionuclides and added tracers, demonstrated a significant exponential relationship between soil loss and tracer loss, with the basic conclusion that ¹³⁷Cs would be a useful tool for measuring erosion.

Caesium-137 is an artificial radionuclide with a half-life of 30.2 years produced by nuclear fission. Widespread global distribution of ¹³⁷Cs into the environment began with atmospheric testing of high-yield atomic weapons in the 1950s and the early 1960s. To a lesser extent atmospheric explosions continued until 1980. The ¹³⁷Cs and other radionuclides were released into the

stratosphere, distributed globally, then moved back to the troposphere and to the earth's surface as fallout, the amount of deposition depending on the atmospheric concentration and rainfall.

Radioactive fallout has been monitored globally since the early 1950s. Strontium-90, identified as the most dangerous radionuclide in fallout due to its biological fixation in bones and the consequent link with bone cancer, was most closely followed, with a map of cumulative deposits published in 1967. Although ^{137}Cs deposition is less well documented, it is possible to construct its deposition pattern from the ^{90}Sr data.

Caesium-137 is by far the most widely used fallout radionuclide in soil erosion and sedimentation research by virtue of its strong adsorption to fine soil particles, a relatively long half-life, ease of measurement and well defined patterns of fallout input. Thus, in agro-ecosystems, its redistribution is a direct indication of erosion, transport and deposition of soil particles occurring during the period extending from the main phase of atmospheric deposition to the time of sampling. Assessment of ^{137}Cs redistribution is commonly based upon comparison of measured inventories (total activity per unit area) at individual sampling points, with an equivalent estimate of the inventory representing the cumulative atmospheric fallout at the site, taking due account of the differing behaviour of cultivated and uncultivated soils. Because direct long-term measurements of atmospheric fallout are rarely available, the cumulative input or reference inventory is usually established by sampling an adjacent undisturbed, putatively uneroded location, generally under permanent pasture, which provides an estimate of total fallout. Where sample inventories are lower than the local reference inventory, loss of caesium labelled soil and therefore erosion may be inferred. Similarly, sample inventories in excess of the reference level are indicative of addition of ^{137}Cs -labelled soil by deposition. The magnitude and direction of the measured deviations from the local reference level provide a qualitative assessment of soil redistribution. In order to derive quantitative estimates of rates of erosion and deposition from ^{137}Cs measurements, it is necessary to establish a relationship between the magnitude of the deviation from the reference inventory and the extent of soil loss or gain. Because empirical calibration data are rarely available, many workers have favoured the use of theoretical relationships or models to provide the necessary calibration function.

Although the basis for the use of the ^{137}Cs technique to document rates and patterns of soil loss is attractive in its simplicity, it is founded on several key assumptions and a number of potential limitations and uncertainties must be recognized and addressed in any application. However, limitations inherent in the use of fallout ^{137}Cs to estimate erosion are no greater than those associated with other techniques used by soil scientists and geomorphologists. If studies are designed to mitigate the effects of these limitations, then measuring the concentration of ^{137}Cs across the landscape can provide reliable and valuable information on erosion and deposition.

A significant expansion in the application of the ^{137}Cs approach has occurred recently, encompassing various locations ranging from glaciated mountain areas in Greenland, through the prairie and steppe regions of Canada and the Russian Federation and semi-arid areas of Spain, to tropical areas of Africa and Thailand. Therefore, the approach must now be seen as having global relevance.

It should be noted, however, that additional inputs of ^{137}Cs fallout to many parts of Europe, associated with the Chernobyl accident in April 1996, have complicated the interpretation of the ^{137}Cs inventories and may render the approach of limited value in such areas.

Although the validity and value of the ^{137}Cs approach has now been demonstrated by numerous studies in a number of environments, considerable scope remains to develop, refine and standardize the procedures employed and to explore additional applications. To date, most of the studies using this approach have been concerned with soil erosion by water, but there is clear potential to extend its application to redistribution of soil by tillage and wind erosion.

The data generated by ^{137}Cs measurements are ideally suited to coupling with GIS and spatial statistics, and for use in verifying computer-generated models for distributed erosion and soil loss. Scope also exists for using estimates of soil loss derived from radiocaesium measurements as a basis for establishing medium-term soil-loss/crop-productivity relationships. In addition, there is potential for extending the use of ^{137}Cs as a sediment tracer from consideration of soil redistribution within individual fields to investigation of the movement and storage of sediment within a drainage basin.

Other fallout radionuclides, including unsupported ^{210}Pb and ^7Be have attracted much less attention, but there is increasing evidence that they offer considerable potential for use in soil-erosion research, individually and complementary to ^{137}Cs . The International Atomic Energy Agency, through several mechanisms, is promoting further applications of fallout radionuclides in soil erosion and sedimentation studies.

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¹³⁷Cs USE IN ESTIMATING SOIL EROSION: 30 YEARS OF RESEARCH

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Abstract

¹³⁷Cs USE IN ESTIMATING SOIL EROSION: 30 YEARS OF RESEARCH

Significant amounts of fallout ¹³⁷Cs from nuclear weapons tests were introduced to the landscape during the 1950s and 1960s. Once ¹³⁷Cs reaches the soil surface it is strongly and quickly adsorbed by clay particles, and is essentially nonexchangeable in most environments. Thus, ¹³⁷Cs becomes an effective tracer of the movement of soil particles across the landscape. Over the past 30 years, researchers have shown that ¹³⁷Cs can be used to study soil movement. Early work used empirical relationships between soil loss and ¹³⁷Cs loss to estimate erosion. This was followed by the development of proportional and theoretical models to relate ¹³⁷Cs movement and soil redistribution. Most of the problems related to the ¹³⁷Cs technique are the same as those encountered with other techniques (i.e., sampling, measurement). The ¹³⁷Cs technique can make actual measurements of soil loss and redeposition in fields, fostering the formulation of better plans to conserve the quality of the landscape. This paper reviews the development of the ¹³⁷Cs technique to show how it can be used to understand erosion and soil movement on the landscape.

1. INTRODUCTION

Soil erosion is a natural process caused by water and wind. The accelerative effects of man's activities on erosion and the off-site damage caused, are major concerns around the world. The size of the problem and concern over degradation of the landscape are well documented [1, 2, 3]. Economic effects of soil erosion along with off-site, downstream damage from eroded soil particles have also been described [3, 4, 5].

Measurements of soil erosion on the landscape using classical erosion techniques are difficult, time consuming, and expensive [6]. Empirical and theoretical mathematical equations/models have been developed. The most widely used is the Universal Soil Loss Equation (USLE), which is an empirical-based equation developed with data collected from soil erosion plots on "typical" soils of the United States east of the Rocky Mountains [7]. The USLE has been used and misused in the United States and around the World [8]. However, it is still the most widely used, powerful and practical tool for estimating sheet and rill erosion on the landscape. A Revised Universal Soil Loss Equation (RUSLE) is currently being used in the United States with applications to a wider range of conditions and locations than the original USLE [9]. There are many other efforts to model soil erosion and its off-site effects [10] that have had varying degrees of success and applications in management and research. One such effort is the Water Erosion Prediction Project (WEPP), which is a process-based, simulation model of soil erosion [11].

Over the past 30 years, researchers have studied the potential of using natural and man-made radioisotopes to study the erosion and sediment-deposition cycle. Several radioisotopes have been used. The potential for using fallout ¹³⁷Cs to provide independent measurements of actual soil erosion rates and patterns and sediment deposition is well documented [12, 13]. The purpose of this paper is to review the development of the ¹³⁷Cs technique for studying erosion and redeposition, based on a bibliography [14] of 1500 papers that show extensive use of the technique globally.

2. BACKGROUND

Most classical methods for estimating soil erosion are based on measuring soil loss from plots or at the edge of a field. They do not give unbiased measurements of actual soil movement, and, more importantly, they do not address spatial patterns of erosion and redeposition within fields. Mathematical models have the same limitations. There is a need to be capable of making measurements at any location on the landscape, especially in areas where other erosion data are not available and where long-term experiments have not, nor cannot be, established. Classical erosion measurement techniques and mathematical models cannot meet these criteria. Tracer techniques have the potential to provide the necessary type of data. However, such techniques can be difficult if tracer must be added to the environment. A tracer is needed that is naturally distributed across the landscape, easily measured, and readily adsorbed to soil particles.

Fallout ^{137}Cs from atmospheric nuclear weapons tests of the 1950s and 1960s is a unique tracer for the erosion and deposition cycle because no natural sources are in the environment. Yet, ^{137}Cs is globally distributed across the earth's surface due to fallout deposition from such tests and releases from nuclear reactors [15, 16]. Before 1952, releases were localized around weapon test sites or reactors. With the coming of high-yield thermonuclear weapons testing in November 1952 [17], ^{137}Cs was injected into the stratosphere and circulated globally [18]. Fallout rates decreased with distance from the northern temperate zone. Regional patterns and rates of fallout were linearly related to precipitation in latitudinal zones [18].

Local variation of fallout ^{137}Cs on the landscape can be significant. Studies have reinforced the need to make local measurements at undisturbed sites rather than by extrapolating from values determined at other locations [19, 20]. The rates of deposition of fallout ^{137}Cs have decreased since the maxima of the early 1960s, and since the mid-1980s have often been below detection limits [15]. Releases from nuclear reactors are usually local in nature. However, the Chernobyl accident in April 1986 caused regional dispersal of measurable ^{137}Cs [21] that affected the total global deposition budget [22]. Thus man's activities related to nuclear energy have distributed a unique radioactive element across the landscape surface in discernible patterns that can be used to trace natural events.

The chemistry of this unique tracer is well understood [18, 23]. Once ^{137}Cs reaches the soil surface it is strongly and quickly adsorbed by clay particles, and is essentially nonexchangeable in most environments [24, 25, 26]. Thus, ^{137}Cs becomes an effective tracer of the movement of surface soil. Distribution of ^{137}Cs in soil profiles at undisturbed sites shows an exponential decrease with depth [27, 28, 29], whereas plowed soils show uniform distribution throughout the plowed layer [30, 31]. Less than 1% of the ^{137}Cs is flushed in solution from a catchment immediately after deposition, and generally less than 0.1% moves in solution per year after the initial flush [32, 33]. Thus most movement of ^{137}Cs across the landscape is due to the physical processes of erosion and sediment deposition.

Accurately measuring ^{137}Cs in environmental samples is easy [34, 61]. In soil erosion studies, the challenge is to elucidate the changing patterns of distribution of ^{137}Cs -tagged soil particles on the landscape. The redistribution of ^{137}Cs between and within landscape elements provides information on soil erosion rates and patterns. Although biological and chemical processes move limited amounts of ^{137}Cs in unique environments, water and wind are the dominant factors moving ^{137}Cs -tagged soil particles between and within compartments of the landscape.

Thus, measurement of ^{137}Cs redistribution on the landscape provides estimates of long-term soil loss. Estimates are location-specific and can be made with minimum disturbance to study sites, giving both spatial patterns and rates of erosion from a single visit.

3. EARLY RESEARCH

The first publication of the use of fallout radioactivity to estimate soil loss was in 1960 by Menzel [35], who measured the transport of fallout ^{90}Sr in runoff from "standard" erosion plots. He concluded that ^{90}Sr loss was greatest from those plots that had the most soil loss. Although this study did not include ^{137}Cs , it showed that the movement of fallout radioactivity on the landscape was related to soil movement.

In 1963, Frere and Roberts [36] measured ^{90}Sr across a small cultivated catchment in Ohio as a function of slope position and shape, and concluded that the pattern was due to the redistribution of soil particles by erosion processes. Graham [37] added ^{85}Sr and ^{131}I to standard erosion plots, and concluded that soil particles affected nuclide removal by runoff water. And Yamagata, Matsuda, and Kodaira [38] concluded that runoff was a factor in the removal of ^{137}Cs and ^{90}Sr from catchments.

In 1965, Rogowski and Tamura published the first of three reports [39, 40, 41] on the movement of ^{137}Cs by runoff, erosion, and filtration from plots at the Oak Ridge National Laboratory in Tennessee, USA. They added ^{137}Cs as a tracer and followed its movement by measuring runoff, soil loss and ^{137}Cs loss at a flume at the end of erosion plots. The first publication [39] was based on the first 83 days of data collection. In follow-up publications in 1970, they discussed the environmental mobility of ^{137}Cs [40] and erosional behavior of ^{137}Cs [41]. They found a significant exponential relationship between soil and ^{137}Cs losses, and concluded that erosion was a major factor in removing ^{137}Cs . Although these studies were not based on fallout nuclear weapons tests, they showed that ^{137}Cs and soil movements were related, and could be used as a tool for estimating soil redistribution on the landscape.

It is interesting that in 1965 Wischmeier and Smith [7] first published the USLE (Universal Soil Loss Equation) in USDA Agriculture Handbook No. 262 on predicting rainfall-erosion losses for cropland east of the Rocky Mountains. The USLE, an empirical equation based on measured soil loss from standard erosion plots on "typical soils" east of the Rocky Mountains in the United States, received wide acceptance in the United States and around the world. The equation is used and misused [8] for areas where the empirical relationship developed for United States soils is probably not applicable. The early work of Rogowski and Tamura on using the ^{137}Cs to measure soil loss, however, has yet to be implemented as a tool for soil conservationists to study soil redistribution.

In another study at Oak Ridge, Dahlman and Auerbach [42] added ^{137}Cs to a grass plot (fescue meadow) and used its redistribution to estimate erosion. They also found an exponential relationship between soil and ^{137}Cs losses. These early studies with high levels of added ^{137}Cs raised the question of whether measurement of the much lower fallout levels could be used to estimate soil redistribution patterns across the landscape.

In 1968, Ritchie and McHenry began a series of studies to determine if fallout levels of ^{137}Cs could be used as a tracer of sediment movement and deposition across natural and agricultural landscapes, supported jointly by the United States Department of Agriculture and the U.S. Atomic Energy Agency (now Department of Energy). In 1970 [43, 44], they concluded that differences in distribution of fallout ^{137}Cs between vegetation types on a catchment in Mississippi were due to soil loss from the landscape. In 1972 [45, 46, 47], they designed an experiment to determine the relationship between losses of soil and of fallout ^{137}Cs in a catchment. Soil loss was estimated using the USLE and fallout ^{137}Cs loss was calculated as a percent compared with a non-eroded reference site. They found an exponential relationship between soil and ^{137}Cs loss, and, along with data from two other catchments [47, 48], concluded that most of the ^{137}Cs loss from the catchments was from the eroded areas.

In 1975, Ritchie and McHenry [49] combined their data with those from the earlier studies of Rogowski and Tamura [39, 40, 41], Menzel [35], Frere and Roberts [36], and Graham [37], and found a significant exponential relationship between soil loss and radionuclide loss. This was encouraging since the data used to develop this empirical relationship came from a variety of soil and erosion conditions, used different radionuclides, and varied from fallout levels to high levels added as tracers. Their basic conclusion was that ^{137}Cs would be a useful tool for measuring soil loss from the landscape.

While most of the early research in the 1970s was in the United States, in 1977 McCallan and Rose [50] used ^{137}Cs and ^{210}Pb to estimate erosion in a basin in Australia. The same year, Wise [51] published a review paper in England on the use ^{137}Cs and ^{210}Pb to measure denudation rates.

Simultaneously, McHenry and Ritchie [52] found that ^{137}Cs distribution in an agricultural field could also be used to show that most of the soil particles were being redeposited within the field rather than being lost from it. This opened a new area of interest in erosion, using the distribution of ^{137}Cs to determine spatial patterns of erosion within a field and areas of net loss and of net gain (deposition) within a landscape element.

In the 1980s, four major centers of research on the use of ^{137}Cs to study erosion were active. McHenry and Ritchie [12] continued their activities in the United States. In Australia, Campbell [53], Elliott [54], Loughran [55], and others used the low levels of ^{137}Cs in the southern hemisphere to measure erosion and sediment deposition. A group lead by de Jong [56] and his students [57, 58] used ^{137}Cs extensively to study erosion on the Canadian prairie. In England, Walling [59, 60, 61] developed a center at Exeter University for using ^{137}Cs to quantify changes in landscape geomorphology. These centers are still active in their research efforts and continue to provide new methods to use ^{137}Cs to quantify soil redistribution across the landscape.

4. EQUATIONS/MODELS

Empirical equations have been developed to explain the relationship between ^{137}Cs and soil loss. The studies vary from simultaneous measurement of losses of soil and of ^{137}Cs (radionuclide) from erosion plots, to correlation between soil loss from the plots and the reduction of ^{137}Cs in these plots, to correlation between estimates of soil loss from fields and the reduction of ^{137}Cs in the soils of these fields. Although important in showing a relationship between soil and ^{137}Cs loss, these studies have many limitations. The general form of these equations is $Y = aX^b$. Such empirical equations are affected by climate, soils, time since fallout, time period of development, and other landscape and environmental factors. Similar concerns for the application and misuse of the empirical-based USLE have been expressed [8]. Empirical equations are applicable only to the data domain used in their development. Using this approach to estimate soil erosion will require the development of an empirical equation (calibration curve) for each site or at best, for each region. Empirical equations are dependent on the time since fallout and time of fallout. While they may help explain and better define the role of different factors that affect the relationship between soil loss and radionuclide loss, the many limitations to their application reduce their usefulness for estimating soil loss on a large scale.

A second approach for using ^{137}Cs to study erosion is to assume that the loss of ^{137}Cs at a site is proportional to the loss of soil. The simplest form of this approach is to equate soil loss to ^{137}Cs loss ($Y = X$), where Y is soil loss in weight per area per time and X is ^{137}Cs loss in percent or weight. However, the X term is usually modified by depth distribution of ^{137}Cs , density of soil, decay corrections, and other coefficients and modifiers [62]. Kachanoski [63, 64] provided an empirical

verification of this approach by measuring ^{137}Cs concentrations in erosion plots at two different times and comparing the results of measured soil loss with measured ^{137}Cs loss. A major assumption with this model is that ^{137}Cs is instantaneously uniformly distributed in the soil profile. Since ^{137}Cs is deposited on the surface and strongly adsorbed, it requires mechanical mixing (plowing) to achieve uniform distribution. Thus, during times of fallout, these conditions are seldom met leading to excess removal of ^{137}Cs with surface erosion, causing overestimation of soil loss. Since fallout and erosion are both related to rainfall, there are concerns about erosion rates during times of heavy fallout. During these times, greater erosion rates would remove proportionately more ^{137}Cs from the surface layer of soil, leading to overestimation of long-term erosion rates. This problem is magnified if erosion rates are higher during maximum fallout, removing a disproportionate amount of the fallout. Usually the proportional method overestimates erosion rates for periods of heavy fallout. The proportional method will probably reflect actual erosion rates for a cultivated area that was undisturbed during the fallout period.

Another approach is theoretical models/mass accounting. Walling and Quine [61] have defined the theoretical models as "the aggregate effect of all redistribution processes operating over the period since the initiation of atmospheric fallout to establish site-specific calibration relationships." Getting all the data needed to run a "process" based model may be a concern in some regions. By determining the factors that influence ^{137}Cs movement, a ^{137}Cs balance for a catchment can be developed and the spatial pattern and loss of ^{137}Cs from the catchment or field can be calculated. Such studies thus allow an understanding of erosion and deposition patterns of soil particles, and have been used [60, 65, 66] to study erosion and deposition patterns on the landscape. These techniques give qualitative and quantitative information on erosion patterns. As with the other techniques, the balance approach requires a determination of the baseline level of ^{137}Cs for the study area. Several studies have cautioned against using fallout measurements to estimate total ^{137}Cs loads in soils of a watershed. Measurements of actual ^{137}Cs should be made at a noneroding site in the catchment. By comparing ^{137}Cs measurement at a study site with the baseline level, one can determine whether erosion (less ^{137}Cs present than at the baseline site) or deposition (more ^{137}Cs present than at the baseline site) has occurred.

While there are limitations to the use of fallout ^{137}Cs to estimate erosion, they are no greater than those associated with other techniques. Campbell et al. [67] suggest that the errors of the ^{137}Cs technique may be less than current techniques used by soil scientists and geomorphologists to study erosion. If we understand the limitations and design studies to reduce the effects of these limitations, then measuring the concentration of ^{137}Cs across the landscape can provide accurate and valuable information on erosion.

5. CONCLUSIONS

Thirty years of research have shown that the distribution of fallout ^{137}Cs on the landscape can be used to measure soil loss. Measurements of spatial patterns of ^{137}Cs can provide unique insights into soil movement and redeposition within a landscape element that other methods cannot provide. The ^{137}Cs method is often the only way to get actual measurements of soil loss and redeposition. As such, research should continue the development of the technique to better understand the changing landscape. Applications should also be encouraged in areas of the world where resources are limited for developing long-term erosion monitoring programs.

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NEW PERSPECTIVES ON THE SOIL EROSION-SOIL QUALITY RELATIONSHIP

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Abstract

NEW PERSPECTIVES ON THE SOIL EROSION-SOIL QUALITY RELATIONSHIP

The redistribution of soil has a profound impact on its quality (defined as its ability to function within its ecosystem and within adjacent ecosystems) and ultimately on its productivity for crop growth. The application of ^{137}Cs -redistribution techniques to the study of erosion has yielded major new insights into the soil erosion-soil quality relationship. In highly mechanized agricultural systems, tillage erosion can be the dominant cause of soil redistribution; in other agroecosystems, wind and water erosion dominate. Each causal factor results in characteristic landscape-scale patterns of redistribution. In landscapes dominated by tillage redistribution, highest losses occur in shoulder positions (those with convex downslope curvatures); in water-erosion-dominated landscapes, highest losses occur where slope gradient and length are at a maximum. Major impacts occur through the loss of organically-enriched surface material and through the incorporation of possibly yield-limiting subsoils into the rooting zone of the soil column. The potential impact of surface soil losses and concomitant subsoil incorporation on productivity may be assessed by examining the pedological nature of the affected soils and their position in the landscape. The development of sound conservation policies requires that the soil erosion-quality relationship be rigorously examined in the full range of pedogenic environments, and future applications of the ^{137}Cs technique hold considerable promise for providing this comprehensive global database.

1. INTRODUCTION

The connection between soil redistribution and soil quality is implicit in many definitions of soil quality. For example, Larson and Pierce [1] define quality as the capacity of a soil to function, both within its ecosystem boundaries (e.g. soil map unit boundaries) and with the environment external to that ecosystem (particularly relative to air and water quality). The basic functions of a soil within the ecosystem are to sustain biological productivity, maintain environmental quality, and to promote plant and animal health [2].

Clearly, soil erosion has a range of ecosystem effects: on the soil itself at the point where erosion occurs; on the landscape where redistribution of soil occurs; and on adjacent ecosystems to which the soil is exported (primarily downstream aquatic ecosystems). Although all three scales of study are important, this review concentrates on the impact of erosion at the pedon and landscape scales; the primary focus is on the new insights into the soil erosion-soil quality relationship that have been gained by application of the ^{137}Cs redistribution technique.

Redistribution alters the chemical, biological, and physical composition of soil at each point in the landscape where it occurs (the pedon scale). These changes in composition may affect the ability of the soil to perform the ecosystem functions outlined above. The literature in soil science and related disciplines abounds in specific examples of the impact of redistribution on soil quality, yet few generalizations have emerged from this voluminous body of work. In most cases, at the point where loss occurs, there will be a decrease in the ability of the soil to function. The impact of soil deposition on function is even less clear: positive and negative impacts on soil quality have been observed. The link between these redistribution-related changes in soil quality and in crop productivity is also elusive.

Commonly, conclusions are based on what should happen to crop yields given the severity of soil-quality changes, rather than on what has been observed to occur.

The redistribution of soil materials also has a profound influence on the spatial pattern of soil quality indicators within the ecosystem boundary (the landscape scale). Redistribution can increase the range of variability within a given landscape unit; however, redistribution also imposes, or reinforces, a distinctive landform-soil property relationship that can be used to stratify landscapes into meaningful response units or experimental units. Hence, although redistribution may increase the overall range of variability, it can also create or exaggerate an overall spatial order within the ecosystem.

Soil transport by redistribution beyond the boundaries of the source ecosystem to adjacent ecosystems (the regional scale) is the final scale of relevance in examining impact on ecosystem function. The sediment itself and the chemical and biochemical components sorbed to it can be significant contributors of soil-derived pollutants to aquatic ecosystems. The effect of soil erosion on air quality, through deflation and transport by wind, can be of importance in specific regions (i.e. the dusts of north-central Africa), but tends to be a short-lived phenomenon elsewhere in the world.

A rigorous evaluation of the causes and impacts of soil erosion at all the scales of relevance is critical for the development of scientifically-sound land-use policies. The need for reliable data upon which to base these planning strategies is starkly illustrated in two recent articles. Pimentel et al. [3] argue that one-third of the world's arable land has been lost to erosion in the past 40 years; the cost of erosion losses in the U.S.A. alone totals \$44 billion per year, and they suggest that these losses could be reduced to a sustainable levels with total expenditures of \$ 8.4 billion per year. In his response to their study, Crosson [4] suggests that their estimate of arable land lost "rests on such thin underpinnings that it cannot be taken seriously" (p. 461). His estimates of the annual cost of erosion-induced productivity losses in the U.S.A. are in the range of \$500 to \$600 million (1986 dollars). Clearly we cannot expect policy-makers to develop sound conservation policies when we are unable to provide a more authoritative data set upon which to base decisions.

2. CHANGING CONCEPTS OF SOIL REDISTRIBUTION

The great contribution of soil-geomorphologists such as R. Ruhe and R. Daniels was to establish that soil redistribution is a natural process that has driven landscape evolution throughout time. The focus of this review is on the increases in soil loss and gain associated with human activity, or, as it is more commonly known, accelerated soil erosion.

2.1. Causes of soil erosion

Traditionally, reviews of the causes of "erosion" have focused almost exclusively on wind and water processes as the major determinants of soil redistribution. The physical processes of wind and water erosion, their relationship to soil and landform properties, the domains in which they operate, and the relative importance of each process in those domains have been exhaustively studied, and many fine reviews exist of this research [e.g. 5].

Recent research on tillage operations [6, 7, 8, 9] has, however, challenged the view that only wind and water erosion need to be examined as the dominant causes of soil redistribution in all landscapes. Working at research sites in agricultural landscapes of Canada and Europe, these authors used a variety of techniques (natural and enriched ¹³⁷Cs redistribution, displacement of simulated clods, simulation modelling) to examine the relative importance of water and tillage erosion. Their

results strongly support the idea that tillage redistribution is a major cause of soil movement in agricultural landscapes.

Cultivation can displace soil upslope or downslope, depending on the direction of the operation; however, because downslope displacement is greater than upslope displacement, a net downslope displacement occurs. The rates of loss attributed to these practices (discussed below) can be high; however, the magnitude of the loss depends on the type and sequence of the tillage operation, as well as on the type of implement and the speed of operation [9].

Clearly, the impact of tillage redistribution is of greatest importance in highly mechanized agricultural systems - in other systems such as "slash and burn" described by Garcia-Oliva et al. [10], other causes of redistribution predominate. The recognition of the relative importance of a given erosion process (water, wind, or tillage) in a specific region is critical for explaining the landscape-scale spatial patterns of loss and gain, as well as for evaluating possible extra-ecosystem impact of soil redistribution.

2.2. Rates of soil redistribution as determined using ^{137}Cs techniques

Observations of rates of soil redistribution are available from a variety of sources. In many cases, they have been made on small research plots to generate data for the development and calibration of predictive erosion models such as the Universal Soil Loss Equation (USLE) or the forthcoming Water Erosion Prediction Project (WEPP). Traditionally, however, these plots were designed to examine only the soil-loss part of the redistribution continuum, hence they are limited in their usefulness when examining redistribution as a whole. Far too often the measured rates of soil loss from these research plots are uncritically extended to the landscape as a whole, without an evaluation of potential redistribution within the source landscape [11].

TABLE I. EXAMPLES OF RECENT VALUES FOR SOIL LOSS

| Location | Field Type | Rate of soil loss in eroded portions of the field (Mg ha ⁻¹ yr ⁻¹) | Reference |
|-------------------------|--------------------------------------|---|-----------|
| Mexico | Pasture and undisturbed forest | 13 | [10] |
| U.K., Belgium | Cultivated | 10 to 20 | [8] |
| Ontario, Canada | Cultivated | 68 to 82 | [9] |
| Saskatchewan, Canada | Cultivated | 20 to 30 | [15, 16] |

The radionuclide ^{137}Cs was deposited on the soil chiefly as nuclear-bomb fallout in the 1950s and 1960s; peak deposition occurred in 1963 in the northern hemisphere, and some areas received more from the Chernobyl disaster in 1986. Upon deposition, ^{137}Cs is tightly bound by colloidal material at the soil surface. Redistribution rates of these mineral- ^{137}Cs complexes is highly correlated with observed rates of soil redistribution [12]. The increasing use of ^{137}Cs is providing valuable sets of observations on redistribution rates within actual agricultural landscapes [13].

The most common research design used in ^{137}Cs studies has been to sample soils along transects or grids. After calculation of the change in ^{137}Cs concentration through time due to redistribution, usually by comparing the levels in the agricultural field to a nearby uneroded or reference site, a value for soil loss or gain at the specific sampling point can be determined. At each study site, a certain proportion of samples will show loss and some may show gain; the overall balance indicates the net export, or, less commonly, import, of soil.

^{137}Cs -derived rates of soil redistribution differ widely among landscapes, regions, and agricultural systems; however, as a generalization for mid-latitude regions, those portions of the landscape where soil loss occurs usually show average rates of 10 to 60 Mg ha^{-1} (Table I); typical average losses are around 15 $\text{t ha}^{-1} \text{ yr}^{-1}$ (although loss at specific points may be an order of magnitude greater). Note, however, that this average figure does not represent soil export from the study site; in many cases, much of the eroded soil is deposited within the source landscape, especially those dominated by tillage erosion.

Deposition rates are considerably more variable than erosion rates. Typically, deposition is concentrated in a small proportion of sampling points within a landscape [14, 15], but rates at specific points may be high.

For large areas of the tropics, including most of Africa and South America, ^{137}Cs results are not available. Lal [5] used existing figures for sediment transport in African river systems to estimate net erosion losses in upstream areas. Most of the African continent shows net erosion losses of < 10 (arid areas) to 50 $\text{Mg ha}^{-1} \text{ yr}^{-1}$. The highest losses occur in the Maghreb region of NW Africa, with losses of $> 75 \text{ Mg ha}^{-1} \text{ yr}^{-1}$.

The rates of redistribution have greater relevance for soil quality if we convert them into depth of soil lost or gained. Using the bulk densities of a surface prairie soil presented in Pennock et al. [16], a 10 $\text{Mg ha}^{-1} \text{ yr}^{-1}$ soil loss from a soil with a bulk density of 1.07 (native prairie) translates to a loss of 0.093 cm yr^{-1} ; the same rate of loss from a soil with a bulk density of 1.42 (a site cultivated for 80 years) corresponds to 0.070 cm yr^{-1} . The loss per year is not dramatic; however, over a period of cultivation, it may be substantial.

The sites discussed above are regionally typical agricultural landscapes, generally excluding representatives of high magnitude-low frequency (or catastrophic) erosion events. Sites affected by extreme events (e.g. high intensity rainfall, major wind erosion events) or locations within a field that have been selectively influenced by erosion processes (e.g. landscape positions with extensive gully) are likely to show much higher rates of loss.

3. SOIL REDISTRIBUTION-SOIL QUALITY INTERACTIONS: THE PEDON SCALE

The impact of redistribution on specific points in the landscape (herein called the pedon scale) must begin with an understanding of the place of that pedon in the overall landscape. The action and interaction of soil processes that result in the properties observed in a given pedon do not occur at

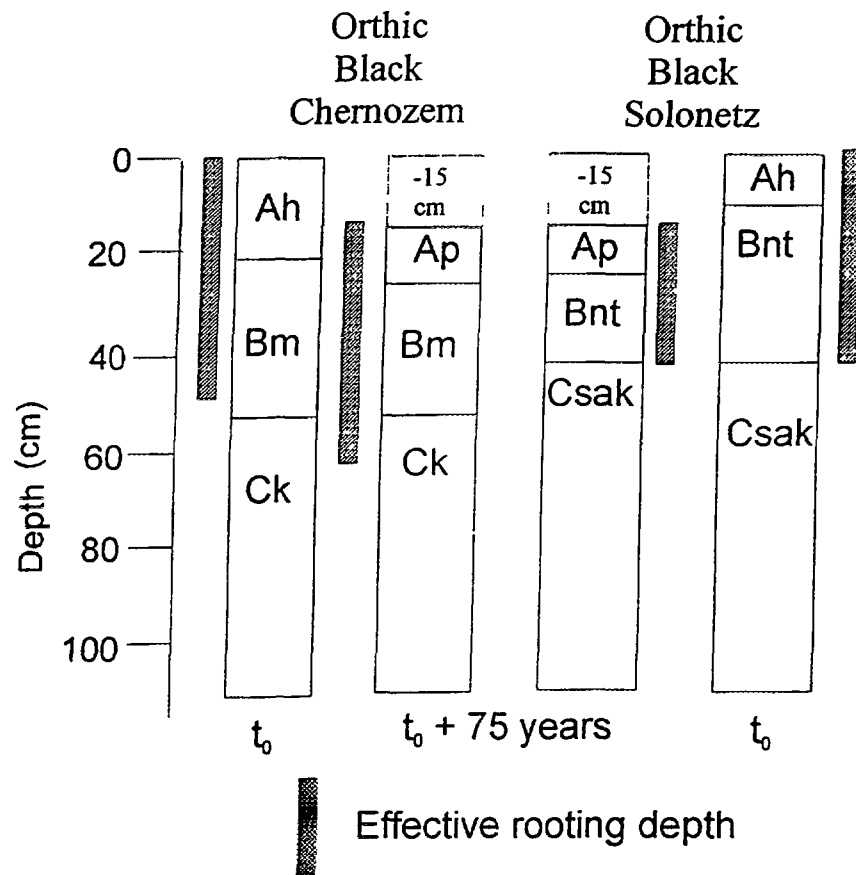


FIG. 1. Schematic diagram showing the impact of 75 years of soil loss and soil gain on the position of the plough layer and the rooting zone within the solum. The changes illustrated are based on annual losses of $20 \text{ t ha}^{-1} \text{ yr}^{-1}$ and equivalent soil gain due to deposition. The depth of cultivation is assumed to be 10 cm, and the rooting zone is 45 cm; both are characteristic of small-grain cropping systems in the Canadian prairies.

random - instead they are responding to a complex set of environmental conditions, which control the type and intensity of the pedogenic processes that occur at any given point in the landscape. The soil and biological processes affecting a given pedon are largely driven by microclimatic differences and the redistribution of water on the soil surface and within the soil. These hydrological and microclimatic differences cause the occurrence of distinctive pedogenic regimes within the landscape, and distinct soils arise in response to these regimes [16].

The need to consider landscape position occurs because of the possibility of confounding landscape effects (i.e. differences in soil properties due to the position of the pedon in the landscape) with erosion effects (i.e. differences in soil properties due to erosion among pedons at the same landscape position). Stone et al. [17] and Daniels et al. [18] examined the confounding influence of landscape position for Ultisols in North Carolina. Overall they found that the most severely eroded soils in the field were usually the least productive to begin with, and that the overall impact of erosion in these landscapes had been previously overestimated by 50%. Hence, we must always ensure that pedons are compared under the same pedogenic regime in the same landscape position, lest the action of erosion be confused with the position of the pedon in the landscape.

The impact of soil redistribution at the pedon scale can be divided into two types (Fig. 1). At sites where erosion is occurring, material is physically removed from the soil surface and transported elsewhere; any layer of fixed thickness within the soil (the cultivation layer, rooting zone) therefore incorporates an equivalent thickness of previously subsoil material. In depositional sites, only one clear impact occurs: the deposited soil buries the previous soil surface and thereby increases the thickness of the uppermost layer in the soil. Each of these three impacts (removal of surface soil material, incorporation of subsoil, deposition of soil) has distinct consequences for soil quality and will be examined separately.

3.1. Consequences of removal of the surface soil material

Erosion physically removes organic material and mineral material from the soil surface. An initial consideration is the potential for selective removal of materials versus bulk removal. In the former, materials are removed in amounts disproportionate to their relative amounts in the bulk soil; in the latter, the eroded material contains the same proportion of organic material and different particle size fractions as the soil from which it was derived. The distinction is important because of the possible occurrence of what has been termed fertility erosion - where erosion selectively removes soil organic matter (SOM) and fine particles (clay, silt), and leaves behind a coarser lag deposit. Because the soil nutrients and exchange sites are concentrated in the SOM and clay fractions, this selective loss of material has greater impact on fertility than the bulk loss would suggest.

Inherent differences in the potential for selective removal can largely be traced to the dominant erosion process. Tillage erosion results from the mechanical displacement of soil-surface material in a net downslope direction; the soil material moves as a mass, and the possibility for selective movement is minimal. In the water erosion process, the possibility for selective transport can occur in interill-dominated systems, but once a critical threshold is reached for a given soil type, detachment and transport is aselective [19]. In rill and gully erosion, detachment and transport is aselective, although separation may occur during deposition. The possibility for selective transport of SOM, fine separates and fine aggregates is perhaps greatest in wind erosion, where, in extreme cases, a coarse sand-gravel lag is left behind on the soil surface [20]. Hence the concept of fertility erosion, although valid in some environments, cannot be uncritically extended to all landscapes.

The loss of SOM from the surface through erosion is probably the single greatest impact of redistribution on soil quality. The use of the ^{137}Cs technique has allowed researchers to apportion the overall losses of SOM [or soil organic carbon (SOC), which is commonly used as a surrogate for SOM] to redistribution causes and to net mineralization causes. For the prairie soils of Canada, de Jong and Kachanoski [21] suggested that, at a range of sites, approximately 50% of the observed SOC losses was due to erosion. Pennock et al. [16] showed that the relative contribution of erosion and mineralization differed depending on the position of the pedon within the landscape. In shoulder positions (those with convex profile, or downslope, slope curvatures), they showed that an overall loss of 64 Mg ha^{-1} of SOC, of an original 117 Mg ha^{-1} , had occurred in 80 years of cultivation; 70% of this was due to redistribution. In footslope positions (those with concave profile curvatures), 45 Mg ha^{-1} of an original 129 Mg ha^{-1} had been lost; 18 Mg ha^{-1} (40%) of this loss was due to redistribution.

The consequences of the loss of SOM on soil quality are considerable. For example, Larson and Pierce [1] suggested the use of pedotransfer functions for soil-quality assessment; SOC appears in five of these (C.E.C., change in SOC itself, bulk density, water retention, and porosity increase). Furthermore, SOM is an important source of nutrients, especially nitrogen, particularly so where fertilizer input is lacking.

3.2. Consequences of the incorporation of subsoil

For layers of fixed thickness in the soil (i.e. rooting zone, depth of cultivation), the loss of material from the soil surface requires the concomitant inclusion of subsoil material of equivalent thickness to the surface losses (Fig. 1). Moreover, it is the nature of the contrast between the surface material and the incorporated subsoil material that determines the severity of the impact that loss has on soil quality and ultimately on soil productivity.

This statement can be illustrated with a simple example (Fig. 2). At the time of initial cultivation, the ploughed layers of an Orthic Black Chernozem (Typic Haploboroll) and of an Orthic Black Solonetz (Typic Natriboroll) were entirely composed of A horizon material (Fig. 1). The rooting layer of the spring wheat in the Chernozem is unconstrained by any impeding layer; in the Solonetzic soil, however, contact with the salt-rich C horizon effectively limits the actual depth of rooting.

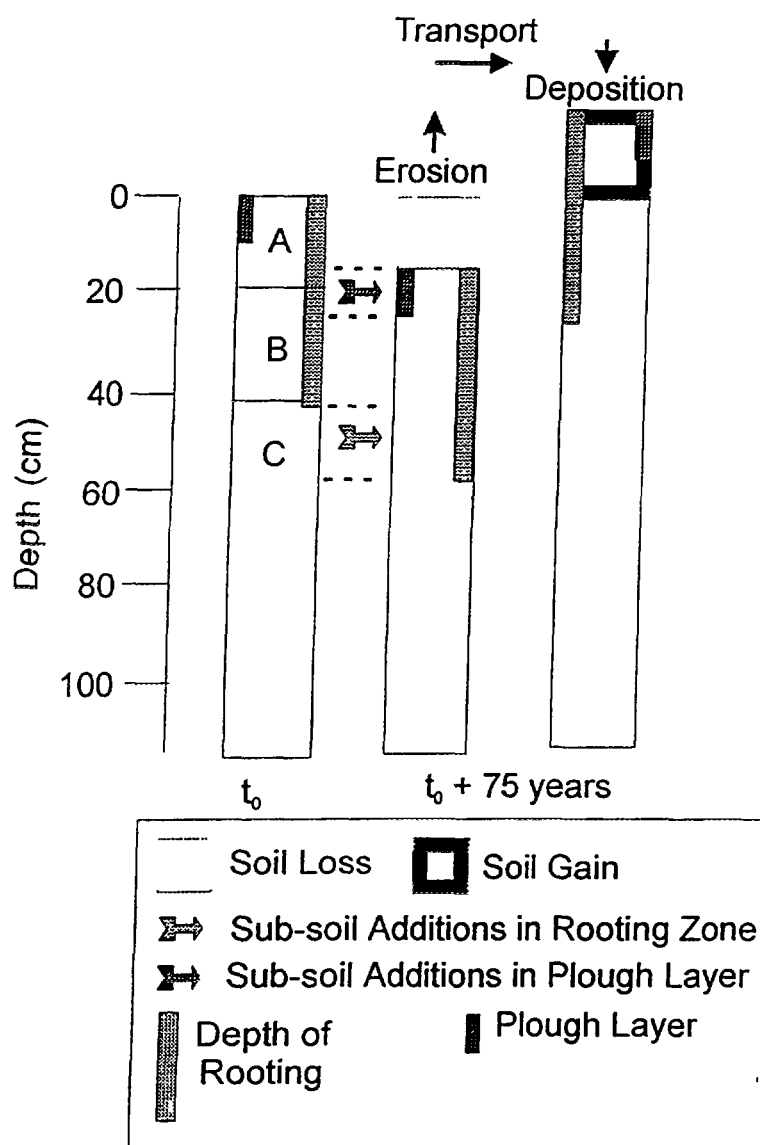


FIG. 2. Schematic diagram showing the differences in the impact of 75 years of soil loss on a Chernozemic (Haploboroll) solum and Solonetzic (Natriboroll) solum (see Fig. 1). The changes shown are based on average losses of $20 \text{ t ha}^{-1} \text{ yr}^{-1}$, a 10-cm cultivation layer, and an optimum rooting zone of 45 cm.

Hence, even at the time of breaking, a minor potential productivity contrast exists between the two soils that is accentuated as erosion proceeds.

For both soils in this example, I assume that erosion is occurring at an equivalent rate of $20 \text{ Mg ha}^{-1} \text{ yr}^{-1}$, which is typical of shoulder (convex downslope) positions in the Canadian prairies [15]. This translates to a loss of approximately 1 cm of soil every 5 years; hence, layers of fixed depth incorporate 1 cm of subsoil every 5 years to balance losses at the soil surface. For the cultivated layer, the incorporation involves mixing of 1 cm of subsoil every year into the remaining 9 cm of Ap; this results in a gradual dilution of the initial surface layer with subsoil.

For the thick Chernozemic soil, even 75 years of erosion does not greatly alter the characteristics of the cultivation layer and the rooting zone (Figs. 1 and 2); the 10-cm thick cultivated layer is still dominantly composed of former Ah horizon material; and the rooting zone is still dominantly within the B horizon. A considerable net export of organic material has occurred; however, the impact of these exports for crop productivity is unlikely to be major, given the lack of change within the rooting zone.

The consequences of erosion losses are much more severe for the Solonchic soil. After 75 years of erosion, the Ap horizon is now found entirely within what was initially the Bnt horizon, although due to dilution and mixing, remnants of the former Ah will be detectable; the rooting zone, assuming it is limited by the salt-rich C horizon, is reduced by 15 cm. The decrease in the thickness of the effective rooting zone, the dense structure of the Bnt, and the deleterious sodium concentrations will all contribute to a major potential decrease in productivity; this decrease will be most apparent in dry years, because the thin rooting layer and lack of root penetration into the peds will greatly limit water uptake by plants.

An equivalent amount of erosion has occurred with both soils. For the Chernozemic soil, the consequences are negligible from a quality perspective, and probably also from a productivity perspective; for the Solonchic soil, however, diminution in soil quality and potential productivity has occurred.

Overall, then, we expect that soils that greatly contrast in quality conditions between the surface soil and the subsoil will have the greatest potential for quality and productivity changes due to erosion; soils with a constant set of properties, or at least a gradual change in levels with depth, will show little or no impact of even substantial amounts of erosion.

The importance of the nature of the subsoil was recognized in the Productivity Index [1], which includes several subsoil properties known to limit root development, and can be used in a range of conditions. Use of the index also allows the vulnerability of a given soil to be assessed [11, 22]. A vulnerability curve, which shows the Productivity Index plotted against surface-soil removal, allows prediction of effects on productivity.

Soils that have a subsoil layer with known growth-limiting soil properties will show the greatest effects of surface loss and subsoil incorporation. As the depth to this layer decreases, the interaction between the layer and plants increases, as does the possibility for productivity limitation.

The soil layers or horizons with the greatest possible impact on productivity, and the reasons for this impact, are shown in Table II. In several of these layers the potential for effect on productivity is clear - in no case does soil quality increase due to the presence of a salt-rich layer or a fragipan in the plough layer or the rooting depth. In the case of plinthite, the layer itself is not limiting when found at depth in the soil; it becomes limiting as soil loss brings it closer to the surface and hardening of the layer to petroplinthite begins [22].

The argillic horizon (those enriched with clay relative to the overlying layer) is a problematic case. The argillic layer is commonly included as a problem subsoil in erosion studies [23]; however,

Stone et al. [17] in their work with Ultisolic soils found that inclusion of the argillic horizon into the cultivated layer due to surface soil loss actually improved overall productivity. They attributed this productivity increase to the higher AWC of these soils (due to higher clay contents); in drier years the higher AWC in turn increased yields over less-eroded landscape positions.

Soils with thin organic-rich layers, e.g. the leaf litter of forest soils or thin A horizons in non-forested soils, are also susceptible to soil quality and productivity changes due to surface soil loss. The example of Garcia-Oliva et al. [10] has been cited above; in the tropical deciduous forest/shifting cultivation agro-ecosystem they examined, the top 4 cm of the soil was critical as a nutrient reservoir. Overall, Lal [5] argues that in agro-ecosystems dominated by tropical soils, the impact of erosion can be especially critical because of the dependence of the agricultural systems on the thin, organically-enriched layer.

3.3. Consequences of deposition of soil

The position in the landscape where deposition occurs varies depending on the nature of the dominant erosional process. In tillage redistribution, the location of deposition is immediately downslope to the point of initiation; the possibility of transport off-site is low, although as Lobb et al. [9] indicated, tillage redistribution can deliver soil to locations where it can subsequently be

TABLE II. HORIZONS AND ASSOCIATED SOIL ORDERS WHERE THE INCORPORATION OF SUBSOIL MATERIAL IS LIKELY TO CAUSE MAJOR CHANGES IN SOIL QUALITY AND PRODUCTIVITY CONDITIONS.

| Horizon | Characteristics and constraints | Soil Orders the horizons are commonly associated with |
|-----------------|---|---|
| Salic | High salt concentration | Aridisols Mollisols |
| Fragic or Duric | Fragipan layer or layer cemented by iron, aluminum or silica; high resistance to root penetration | Spodosols Alfisols Ultisols Inceptisols |
| Natric | High sodium concentration and dense structure | Mollisols |
| Plinthite | High iron and aluminum oxide content; hardens upon drying; high resistance to root penetration | Oxisols Ultisols |
| Argillic | Increase in clay content relative to overlying soil; increases in root penetration resistance | Alfisols Ultisols |
| Oxic | High possible Al^{3+} concentrations in low-pH conditions | Oxisols |
| Spodic | High Al^{3+} or metal levels high in low-pH conditions | Spodosols |

removed by channelized flow. Deposition of soil from the wind stream most commonly occurs immediately downwind of the source of the soil; only the finest particles can be carried for considerable distances in the windstream [20].

The greatest potential for off-site impact occurs due to soil transport by channelized flow of water. Deposition of sediment transported by flowing water occurs where the energy available for transport decreases due to a decrease in gradient (a decrease in slope), a decrease in depth (e.g. where a channel widens or becomes unconfined), or an increase in surface roughness (e.g. where it encounters vegetation). Hence, deposition will occur within the field if one of those conditions is met; if the channel exists, then the soil may be transported out of the field to downslope water bodies.

The on-site effects of deposition (i.e. within the field where initiation of soil movement began) on soil quality are rarely of great consequence. A significant short-term impact can be the burial of seeds or seedlings, and hence a decrease in yield in that year. On-site deposition leads to a thickening of the soil surface at the point of deposition leading to the development of cumelic soils (Fig. 1). Pennock et al. [16] found that deposition of soil improved almost all the indicators of soil quality in a Boroll landscape in Saskatchewan. Again, however, generalization is risky - if the soil delivered from upslope is of significantly lower quality than the original soil, a decrease in quality of the surface layer would occur.

4. SOIL QUALITY-SOIL REDISTRIBUTION INTERACTIONS: THE LANDSCAPE SCALE

In most non-level, cultivated landscapes, the spatial pattern of soil quality indicators at the surface is predominantly controlled by redistribution. Erosion and deposition are associated with distinctive segments of the landscape; in turn, the gain (or loss) of soil causes the occurrence of a distinctive suite of soil properties at the landscape segments with which they are associated.

As with soil deposition, the spatial pattern of redistribution in landscapes depends heavily on the nature of the dominant erosion process. Wind erosion is the only process that can remove significant amounts of soil from level landscapes, and there are examples of extremely high rates of such loss [24]. The highest losses due to water erosion occur in landscape positions where channelized flow (rills, gullies) occurs in the field. These positions are often concave in across-slope (or plan) curvature and have large, upslope contributing or catchment areas. Soil loss due to tillage is highest where the profile curvature of the slope is convex [6]. The difference between the position of highest soil loss due to channelized flow and due to tillage was used by Govers et al. [8] to assess the relative efficacy of the two processes in their study landscapes.

One of the consequences of the widespread use of ^{137}Cs redistribution has been the emergence of a well documented spatial pattern of soil redistribution in landscapes [8, 14, 15, 25]. Slope segments with convex profile curvatures (hereafter called shoulder elements [26]) experience the highest rates of loss; the segments with concave profile curvatures (footslopes) experience net soil gain. The role of the mid-slope or backslope elements (i.e. lacking significant plan or profile curvature) differs; for tillage erosion these elements are probably dominated by transport of soil from upslope, although infilling of small depressions in the slope will occur; if sufficient water velocity or depth exists, they can be areas of significant soil loss in a water-erosion dominated system [25].

The chronosequence of soil studied by Pennock et al. [17] allowed us to trace the evolution of this distinctive pattern. We examined soil redistribution using ^{137}Cs at four sites on a glacial till surface in Saskatchewan, Canada: a native site and sites with 12, 22 and 80 years of cultivation. In the native site, the distribution of ^{137}Cs was random and showed no association with landform [17]. After breaking of the land and 12 years of cultivation, a chaotic pattern of loss and gain occurred,

TABLE III: SUMMARY OF SELECTED SOIL QUALITY INDICATORS AT A NATIVE SITE AND AFTER 80 YEARS OF CULTIVATION IN A BOROLL DOMINATED LANDSCAPE IN SOUTHERN SASKATCHEWAN, CANADA [16]

| | Shoulder elements | | Low catchment area footslope elements | | High catchment area footslopes and level depressional elements* | |
|--|-------------------|------------|--|------------|---|------------|
| | Native site | 80-yr site | Native site | 80-yr site | 12-yr site | 80-yr site |
| pH (0-15 cm) | 7.4 | 7.9 | 6.6 | 7.6 | 7.2 | 7.7 |
| Bulk density (g cm ⁻³) | 1.07 | 1.42 | 1.01 | 1.33 | 1.39 | 1.4 |
| Ah/Ap thickness (cm) | 14 | 12 | 22 | 15 | 11 | 26 |
| Total N (Mg ha ⁻¹) | 10.8 | 6.2 | 11.6 | 8.0 | 4.9 | 9.1 |
| SOC (Mg ha ⁻¹) | 117 | 53 | 129 | 84 | 57 | 113 |

*No native sites were sampled in this area and comparisons are based on the site with 12 years of cultivation

again with no discernable relationship to landform. After 22 years, however, a clear redistribution-landform relationship emerged - high soil loss in the shoulder positions, and deposition in both the low-catchment area and high-catchment footslopes. After 80 years of cultivation, the soil had been scoured out of the low catchment area footslopes and deposited only in high-catchment footslope areas.

We also examined the impact of redistribution on soil quality, and the importance of redistribution for the current pattern of quality indicators was clear (Table III). For soil pH and bulk density of the 0- to 15-cm layer, redistribution has narrowed the overall variability at the site. In both cases, the levels are approaching the values of the subsoil, indicating either the incorporation of subsoil or deposition of subsoil materials from upslope. The A-horizon differences illustrate further the problems with the use of topsoil discussed above: the naturally thin A horizons in the shoulders show little net loss, but the thicker A horizons in the footslopes show considerable losses; yet the highest ¹³⁷Cs losses have occurred in the shoulders. The A horizon cannot, however, drop below the cultivation depth (approximately 12 cm at this site), hence it cannot be used as a reliable indicator of absolute soil loss.

Considerable losses of soil N and SOC have occurred in the shoulder and low-catchment area footslope elements; considerable gains of these properties, and of A-horizon material generally, have occurred in the high-catchment area footslope positions. The impact of redistribution has been to reverse the spatial pattern of these properties found at the native site, where shallow-rooted wetland

vegetation and anaerobic conditions originally limited biological productivity in the high-catchment area footslope areas. Hence the current pattern of soil-quality indicators at least in the surface increment of the soil, strongly reflects the action of redistribution.

5. CONCLUSIONS AND PROGNOSSES

Recent developments in soil-redistribution research, such as the use of ^{137}Cs and the increasing evidence of the importance of tillage, have provided new perspectives to examine the soil quality-soil redistribution relationship. Clearly, soils show a range of vulnerability to loss; curiously, however, much of the research has been concentrated on soils such as the Mollisols, whose vulnerability levels are, overall, probably quite low. Hopefully the wider application of relatively inexpensive techniques such as the ^{137}Cs approach will allow the quality-redistribution relationship to be assessed on soils with higher vulnerability to redistribution, especially those in tropical developing countries.

A dichotomy also occurs between the developed and developing worlds in terms of the prognosis for redistribution itself. In many areas of the developed world, the increased use of herbicide-based weed control has led to increasing adoption of reduced tillage or no-till cultivation systems. Regardless of the cause of erosion, reductions in tillage and increases in the residue cover following field operations will slow the rate of soil redistribution; in a tillage-redistribution dominated system, it may almost eliminate accelerated erosion.

A positive prognosis does not, however, exist for many areas of the developing world. Population pressures in many developing countries may lead to increased utilization of marginal land and to greater use of mechanized tillage systems; both may lead to increases in accelerated erosion. Hence, while the principles of effective erosion control are well understood and are being applied in many developed countries, the socio-economic constraints inherent in the agricultural systems of many developing countries is likely to limit the adoption of these vital control principles.

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DEPOSITION, TRANSFER AND MIGRATION OF ^{137}Cs AND ^{90}Sr IN SWEDISH AGRICULTURAL ENVIRONMENTS, AND USE OF ^{137}Cs FOR EROSION STUDIES

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Abstract

DEPOSITION, TRANSFER AND MIGRATION OF ^{137}Cs AND ^{90}Sr IN SWEDISH AGRICULTURAL ENVIRONMENTS, AND USE OF ^{137}Cs FOR EROSION STUDIES.

Intensive atmospheric tests of nuclear bombs in the late 1950s and early 1960s resulted in fallout on Sweden of 3 kBq/m^2 ^{137}Cs and 2 kBq/m^2 ^{90}Sr . To determine how soil characteristics influence radionuclide transfer to red clover, pot experiments with ^{137}Cs and ^{90}Sr were made with 178 Swedish mineral soils; significant negative correlations were obtained with levels of P, K and Ca. To quantify impact in field conditions, experiments with artificial depositions on microplots were started in 1961. Transfer to barley on 12 topsoils combined with sandy and clay subsoils, and to grass on two contrasting pastures, was followed over two decades. The subsoil type was found to be important.

Fallout of ^{137}Cs from the Chernobyl accident, up to 200 kBq/m^2 in some areas of Sweden, was studied between 1986 and 1994. As in the microplot experiments, transfer of ^{137}Cs to grass was higher than to arable crops. Transfer rates were high in the first year(s), and then decreased differently from year to year, as shown by a new Tar (i.e. annual reduction rate in nuclide transfer) concept. The use of Chernobyl fallout to investigate soil redistribution on arable fields is briefly discussed; a frame-scraper soil-sampling procedure is proposed to improve the accuracy and precision of the ^{137}Cs technique for evaluation of soil erosion.

1. INTRODUCTION

With the advent of the atomic age, radiotracers became a common research tool. For agricultural scientists, isotopic labelling provided the ability to follow the fate of applied fertilizers in soil-plant systems. Tracer techniques have been used in root studies to determine uptake of nutrients and water from different depths in the soil profile [1], and to elucidate the fate of pesticides and other pollutants.

The release of fission products and transuranic elements by detonated nuclear bombs and nuclear accidents, has stimulated research on the behaviour of radionuclides in terrestrial and aquatic ecosystems. In Sweden, extensive pot experiments and long-term field experiments have been carried out with the long-lived fission radionuclides ^{137}Cs and ^{90}Sr . After the Chernobyl accident, field surveys and "real" field studies on the behaviour of, especially, ^{137}Cs were carried out [2].

In this paper, we briefly review these investigations, especially transfer and migration of ^{137}Cs , and, for comparison, ^{90}Sr , under Swedish field conditions. Some aspects of the use of ^{137}Cs in soil-erosion studies are considered. A frame procedure for soil sampling is described, for use in a pilot study for calibration of the ^{137}Cs -technique on a silty soil within a Chernobyl-fallout area.

Deposition of Chernobyl-Caesium in the Nordic countries

Source: Nordic Radioecology
F.H.H. Dahlgaard, Helsinki 1994

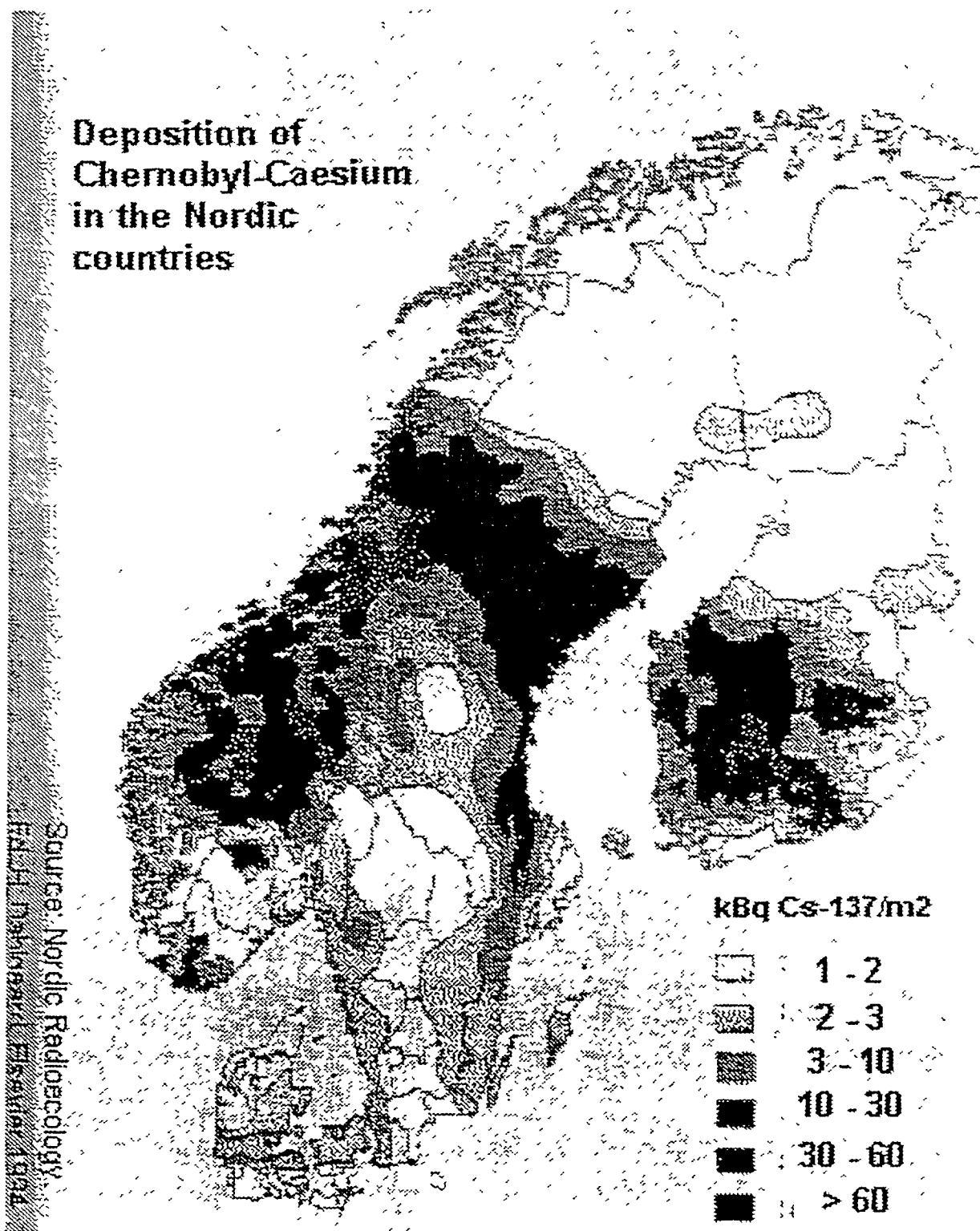


FIG. 1. Ground deposition of ¹³⁷Cs from Chernobyl, kBq/m², in nordic countries Denmark, Finland, Norway and Sweden (T. Selnäs in Dahlgaard, 1994).

2. RADIOACTIVE FALLOUT IN SWEDEN

2.1. Nuclear bomb tests

Tests of thermonuclear weapons have been conducted since 1945. Intensive tests during 1954-58 and 1961-62 resulted in radionuclide contamination of the atmosphere that persisted globally for several years. In the northern hemisphere, mean fallout values are estimated at 2.8 kBq/m² of ¹³⁷Cs and 2 kBq/m² of ⁹⁰Sr [3]. The mean deposition on Sweden from the test at Novaja Zemlja alone was estimated at 3 kBq/m² of ¹³⁷Cs [4].

2.2. The Chernobyl accident

On 26 April 1986, two explosions occurred in a reactor at the Chernobyl power plant, situated 120 km north of Kiev in the Ukraine. The core was exposed, with release of large amounts of fuel particles containing fission nuclides and transuranics. Radionuclides associated with larger particles were deposited near the site, whereas smaller particles (approximately 1 mm) were, with the heat of the explosion, lifted to an altitude of > 1,000 m and carried away in the wind.

During April 26-30 and May 8-10, wind direction was towards Scandinavia. The heaviest fallout occurred during the first phase, estimated to range from 10 to > 80 kBq/m² in some areas of north and mid Sweden, with maxima in the Gävle community of > 200 kBq/m² (Fig. 1). The inventory of ¹³⁷Cs for Sweden as a whole was about 4.25 x 10¹² kBq, about 5 % of the total release from the Chernobyl reactor [4]. It is notable that while the ratio of ¹³⁷Cs / ⁹⁰Sr was about 1.5 in bomb-test fallout, deposition from Chernobyl contained only about 1 % of ⁹⁰Sr compared to ¹³⁷Cs. The latter also contained ¹³⁴Cs in amounts that give a ¹³⁴Cs / ¹³⁷Cs ratio of 1.6 [4].

3. SOIL-PLANT TRANSFER OF ¹³⁷Cs AND ⁹⁰Sr IN POT EXPERIMENTS

In the late 1950s and early 1960s, pot experiments using 178 Swedish mineral soils were carried out at the State Experimental Research Institute, Uppsala. The aim was to identify soil characteristics that influence the transfer of ¹³⁷Cs and ⁹⁰Sr from soil to red clover [5, 6]. With the aid of multiple regression analysis and the equation $Y = A * X_1^{b_1} * X_2^{b_2} * \dots * X_n^{b_n}$, (where Y is the transfer unit, A is the intercept, X₁, X₂ X_n are soil factors and b₁, b₂ b_n are coefficients to be estimated), significant correlations were found for transfer of ¹³⁷Cs and ⁹⁰Sr with soil factors.

The transfer of ¹³⁷Cs to red clover was negatively correlated with exchangeable potassium (K_{AL}) and non-exchangeable potassium (K_{RES}) (Fig. 2), and with exchangeable calcium (Ca_{AL}) (R² = 0.35). The transfer of ⁹⁰Sr was found to be negatively correlated also with exchangeable calcium (Ca_{AL}), soil pH, cation exchange capacity (CEC) and labile phosphorus (P_{AL}) (R² = 0.87).

4. SOIL-PLANT TRANSFER OF ¹³⁷Cs and ⁹⁰Sr UNDER FIELD CONDITIONS

Comparisons of radionuclide transfer to different field-grown crops are meaningless without referencing plant activity to total inventory in the soil. This concept [7], and its TFG-value [8], has been used at the Department of Radioecology, Uppsala, in field microplot studies with nuclides experimentally applied to the plough layer of arable soils and to the surface of grassland sites:

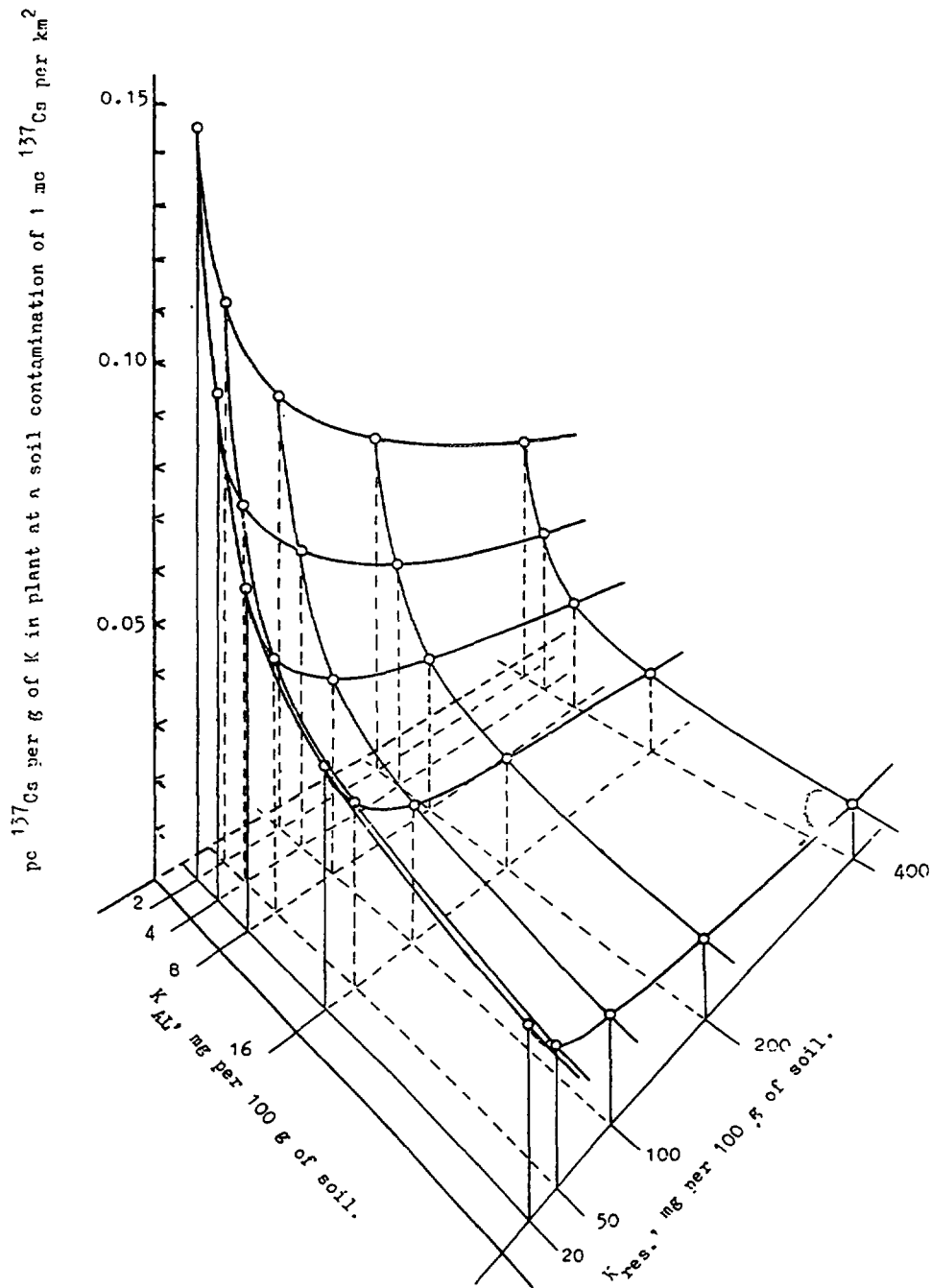


FIG. 2. Plant absorption of ^{137}Cs from soils differing in content of exchangeable potassium (K_{AL}) and residual potassium (K_{RES}). The diagram is based on data obtained in pot experiments with 178 Swedish soils.

$$\text{TFg} = \frac{\text{Activity in plant (Bq kg}^{-1} \text{ dry wt.)}}{\text{Total deposition (Bq m}^{-2})} \quad (1)$$

The TFg, recalculated to the time of contamination, is expressed as the fraction of activity per kg dry matter recovered from a deposition of 1 Bq/m².

Decreasing rates of transfer to crops, independent of physical decay, are observed in most terrestrial environments due to declining availability. An ecological half-life, T_{ec} , similar to that of radioactive decay, can be calculated from the general exponential equation:

$$Y = Y_0 * \exp(A*t) \quad (2)$$

As found in pasture experiments with ^{137}Cs after the Chernobyl accident [2], the annual reduction in nuclide transfer was not exponential (A is not a constant in Eq. 2) on grassland sites, at least not during a lag period after deposition. Instead an equation of the type

$$f(t) = \exp(A*t^N + B) \quad (3)$$

can be used [9], where A, N and B are constants and t is time (years). The constants are obtained by a least square fit of observed annual mean TFg values. From Eq. 3 an inverse, denoted **Tar**, can be calculated and used as an expression for the annual reduction rate in nuclide transfer. Contrary to the **Tec** concept, the **Tar** concept depends only on the real reduction in nuclide transfer from year to year.

4.1. Microplot experiments

4.1.1. Transfer to grass

In April 1961 two contrasting types of Swedish pastures were artificially contaminated with ^{90}Sr and ^{137}Cs [10]. One was a permanent pasture on a sandy soil and the other a grazing ley on a clay soil. Results of the two experiments for the period 1961-1981 are shown in Table I.

There was more transfer on the permanent pasture than on the grazing ley. Even after 20 years, the nuclides were retained in the soil and were available for uptake. The TFg values varied from year to year due to unequal rooting, and showed a decreasing trend over time. Although there was more leaching of ^{90}Sr than of ^{137}Cs , the transfer of ^{90}Sr was higher than that of ^{137}Cs .

4.1.2. Transfer to barley

Soil from the plough layer (0-20 cm) of 12 Swedish topsoils were homogenously contaminated with carrier-free solutions of ^{137}Cs and ^{90}Sr , and placed on plots on two contrasting subsoils, a sandy subsoil of 1-m depth and a clay subsoil [11]. Results are shown for barley grain in Table II for the period 1961-80.

The TFg values were 5- to 10-fold higher for ^{90}Sr than for ^{137}Cs . For both nuclides, TFg varied with year due to unequal release of K and Ca from the uncontaminated subsoils [1]. For the same reason, the long-term trends also differed between ^{137}Cs and ^{90}Sr . As regards the influence of top-soil characteristics, the TFg for ^{137}Cs decreased with increasing contents of clay and exchangeable K, but increased with increasing contents of organic matter. The Tfg for ^{90}Sr decreased with increasing pH and CEC, and with exchangeable contents of Ca and P. The results were thus consistent with those from the pot experiments described above.

5. MIGRATION of ^{137}Cs AND ^{90}Sr IN SOIL PROFILES

5.1. Microplot experiments

5.1.1. Pasture experiments

Leaching of the two radionuclides was determined in 1967 in soil profiles at both pasture sites, and was repeated in 1981 at one of the sites [10]. As expected, ^{90}Sr was more mobile than ^{137}Cs (Tables III and IV). No large effect of fertilization was observed, except for one of the NPK-

treatments on the permanent pasture/sandy soil, where acidification by annual application of ammonium sulfate increased leaching of ^{90}Sr .

TABLE I. MEANS OF TFg OF ^{137}Cs AND ^{90}Sr IN GRASS ON PERMANENT PASTURE/SANDY SOIL AND ON GRAZING LEY/CLAY SOIL

| Year | Permanent pasture sandy soil | | Grazing ley clay soil | |
|--|------------------------------------|------------------|-----------------------------|------------------|
| | ^{137}Cs | ^{90}Sr | ^{137}Cs | ^{90}Sr |
| (m ² kg ⁻¹ DW x 10 ⁻³) | | | | |
| 1961 | 22.2 | 73.0 | 9.25 | 23.2 |
| 1962 | 6.85 | 50.4 | 0.71 | 15.1 |
| 1963 | 3.48 | 33.0 | 0.41 | 11.8 |
| 1964 | 31.1 ^a | 24.8 | 7.93 ^a | 9.9 |
| 1965 | 15.3 | 21.9 | 1.37 | 9.7 |
| 1966 | 7.16 | 18.7 | 0.80 | 8.7 |
| 1967 | 2.48 | 18.4 | 0.35 | 6.6 |
| 1968 | 1.86 | 18.3 | 0.24 | 6.1 |
| 1969 | 1.10 | 13.7 | 0.10 | 5.1 |
| 1970 | 0.68 | 15.3 | 0.11 | 5.5 |
| 1971 | 0.83 | 14.6 | 0.18 | 5.7 |
| 1972 | 0.96 | 10.9 | 0.18 | 5.4 |
| 1973 | 0.70 | 11.8 | 0.10 | 3.8 |
| 1974 | 0.74 | 12.8 | 0.22 | 4.5 |
| 1975 | 0.72 | 13.7 | 0.08 | 4.2 |
| 1976 | 0.78 | 10.8 | 0.06 | 2.8 |
| 1977 | 0.67 | 10.3 | 0.06 | 3.6 |
| 1978 | 0.82 | 12.2 | 0.13 | 3.5 |
| 1979 | 0.82 | 12.0 | 0.10 | 3.5 |
| 1980 | 0.53 | 12.6 | 0.12 | 4.8 |
| 1981 | 0.44 | 10.7 | 0.10 | 3.6 |
| 72-81 | 0.72 | 11.8 | 0.12 | 4.0 |

^aRecontaminated in 1964.

5.1.2. Lysimeter experiment

Migration in soil profiles of ^{137}Cs and ^{90}Sr was determined with a lysimeter experiment started in 1962 [13], with three topsoils placed on a sandy subsoil and a clay subsoil (Table V). The percentage of activity migrating from the three plough layers to the two subsoils over 20 years was higher for ^{90}Sr than for ^{137}Cs . The migration of ^{90}Sr was greater to the clay subsoil than to the sandy subsoil; the difference may depend on deeper rooting in the former than in the latter subsoil. There was no consistent effect of applying K-fertilizer at 124 kg K/ha/yr.

TABLE II. TFg OF ^{137}Cs AND ^{90}Sr IN BARLEY GRAIN, AVERAGED OVER 12 TOP-SOILS, ON SUBSOIL I (SAND) AND II (CLAY)

| Year | ^{137}Cs | | ^{90}Sr | |
|--------------|--|--------------|------------------|-------------|
| | I | II | I | II |
| | (m ² kg ⁻¹ DW x 10 ⁻³) | | | |
| 1961 | 0.030 | 0.054 | 0.58 | 0.46 |
| 1962 | 0.038 | 0.044 | 0.27 | 0.21 |
| 1963 | 0.050 | 0.065 | 0.56 | 0.46 |
| 1964 | 0.028 | 0.045 | 0.33 | 0.28 |
| 1965 | 0.041 | 0.057 | 0.42 | 0.32 |
| 1966 | 0.048 | 0.074 | 0.33 | 0.32 |
| 1967 | 0.035 | 0.063 | 0.18 | 0.10 |
| 1968 | 0.023 | 0.032 | 0.34 | 0.22 |
| 61-68 | 0.036 | 0.054 | 0.38 | 0.30 |
| 1969 | 0.047 | 0.042 | 0.48 | 0.17 |
| 1970 | 0.066 | 0.053 | 0.52 | 0.25 |
| 1971 | 0.047 | 0.036 | 0.47 | 0.22 |
| 1972 | 0.046 | 0.048 | 0.55 | 0.19 |
| 69-72 | 0.052 | 0.045 | 0.50 | 0.21 |
| 1974 | 0.066 | 0.030 | 0.54 | 0.28 |
| 1975 | 0.058 | 0.034 | 0.40 | 0.19 |
| 1976 | 0.045 | 0.028 | 0.32 | 0.15 |
| 1977 | 0.034 | 0.024 | 0.54 | 0.23 |
| 1978 | 0.024 | 0.019 | 0.52 | 0.23 |
| 1979 | 0.028 | 0.022 | 0.41 | 0.21 |
| 1980 | 0.029 | 0.023 | 0.44 | 0.22 |
| 74-80 | 0.041 | 0.042 | 0.45 | 0.22 |
| 61-80 | 0.041 | 0.042 | 0.43 | 0.25 |

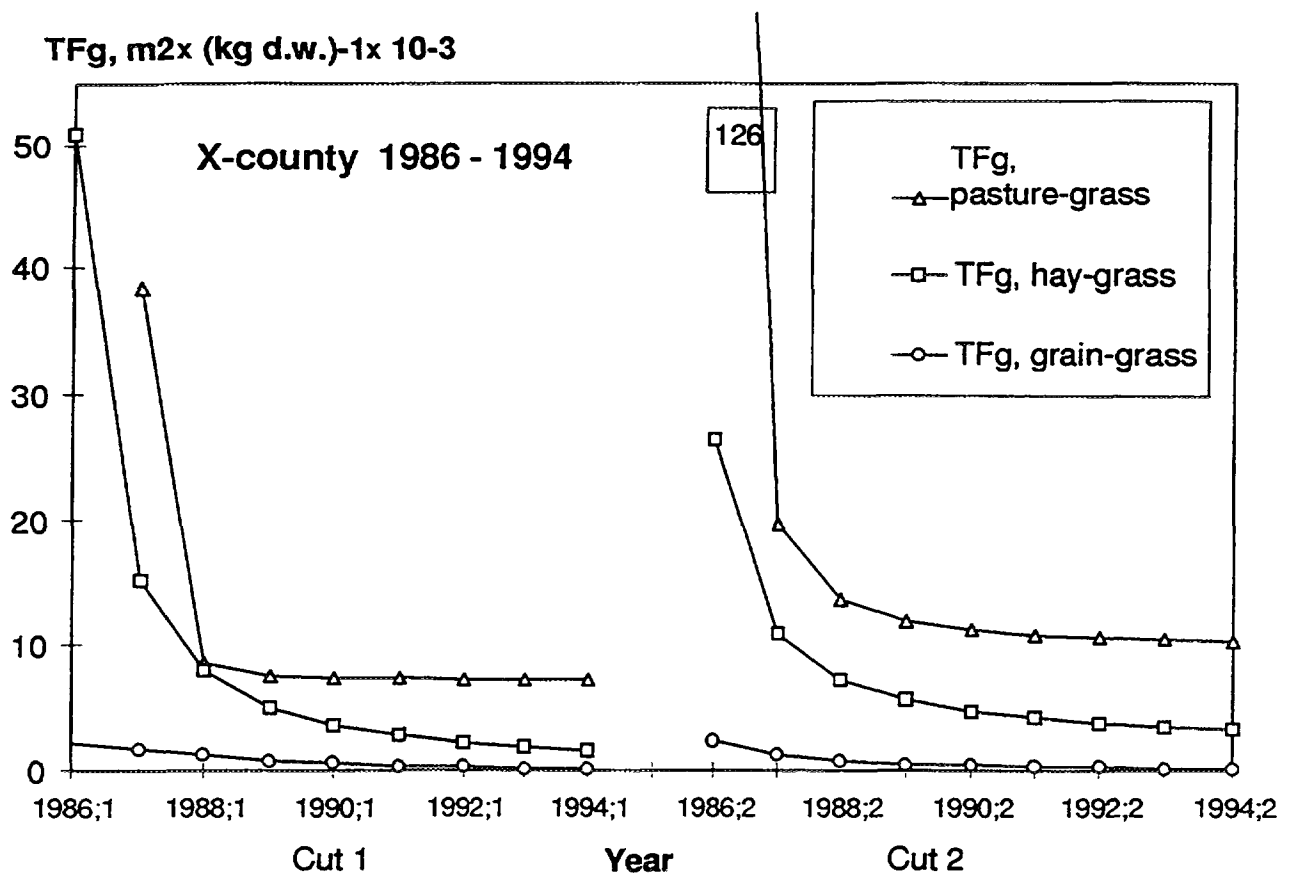


FIG. 3. Annual changes in the X-county of ¹³⁷Cs transfer to two cuts of pasture grass, hay grass and grain-grass. Mean values for this crop have been used for a least square fit according to Eq.3.

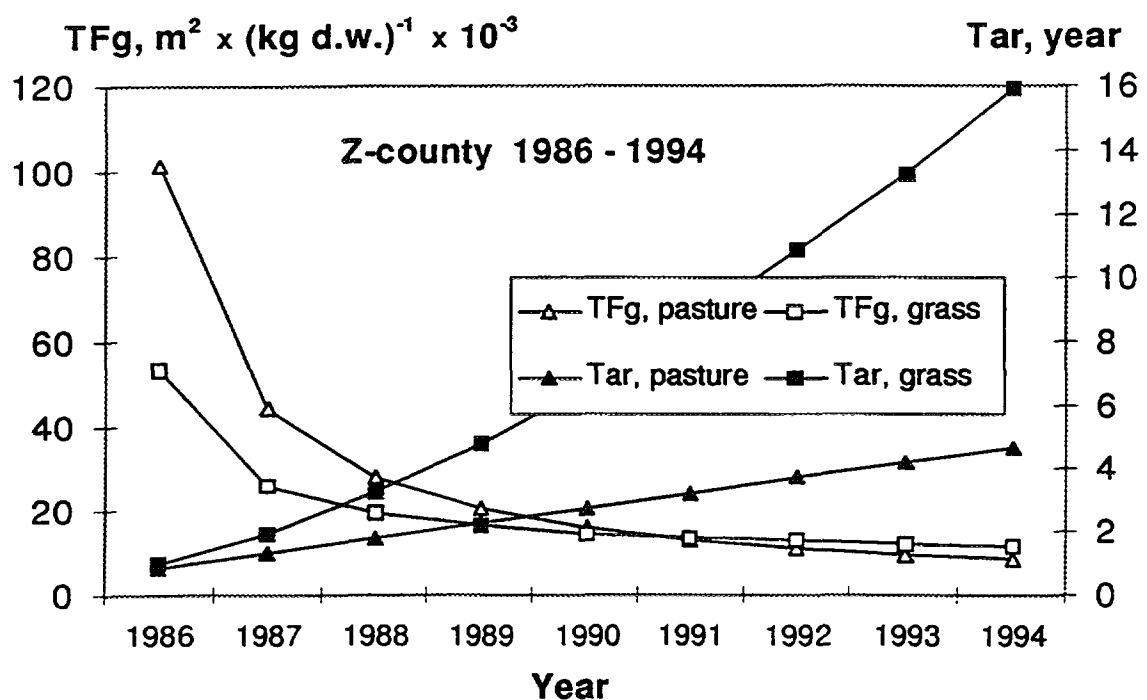


FIG. 4. Annual change in the Z-county of ¹³⁷Cs transfer to pasture and hay grass. Mean values for this crop have been used for a least square fit according to Eq. 3. The Tar-values, as calculated from this equation, show the change in half-time of ¹³⁷Cs transfer with year.

TABLE III. MEDIAN DEPTH OF ACTIVITY IN THE SOIL PROFILE
ON PERMANENT PASTURE/SANDY SOIL IN 1967.

| Nuclide | Range for NPK treatments | Range for lime treatments | Mean |
|-------------------|-----------------------------|------------------------------|------|
| | (cm) | | |
| ¹³⁷ Cs | 1.7 - 2.6 | 1.8 - 2.1 | 1.9 |
| ⁹⁰ Sr | 3.8 - 10.3 | 4.1 - 5.7 | 5.6 |

TABLE IV. MEDIAN DEPTH OF ACTIVITY IN THE SOIL PROFILE
ON GRAZING LEY/CLAY SOIL IN 1967 AND IN 1981.

| Nuclide | Year | Range for PK treatments | Range for N treatments | Mean |
|-------------------|------|----------------------------|---------------------------|------|
| | | (cm) | | |
| ¹³⁷ Cs | 1967 | 3.0 - 3.2 | 2.6 - 3.8 | 3.0 |
| | 1981 | 5.0 - 5.7 | 5.0 - 5.6 | 5.3 |
| ⁹⁰ Sr | 1967 | 5.0 - 5.7 | 5.2 - 5.3 | 5.2 |
| | 1981 | 8.4 - 10.2 | 9.0 - 9.2 | 9.1 |

5.2. After Chernobyl

The leaching of ¹³⁷Cs was also determined in soil profiles at Swedish sites affected by fallout from Chernobyl [12]. Results from four of these sites are shown in Table VI. The radionuclide migrated downwards more in the peat soil than in the three mineral soils. However, most of the ¹³⁷Cs remained in the surface 5 cm even after 8 years.

TABLE V. FRACTION OF TOTAL ACTIVITY TRANSFERRED FROM THE TOPSOIL TO THE SUBSOIL DURING 20 YEARS

| Subsoil type | Topsoil type | ¹³⁷ Cs, NO ₃ -N | | ⁹⁰ Sr, NH ₄ -N | |
|--------------|--------------|---------------------------------------|-----------------|--------------------------------------|------|
| | | -K ^a | +K ^b | -K | +K |
| | | (%) | | | |
| Sand | Sand | 0.41 | 0.08 | 8.8 | 15.6 |
| | Silt | 0.25 | 0.13 | 5.6 | 7.1 |
| | Clay | 0.18 | 0.20 | 6.2 | 7.2 |
| Clay | Clay | 0.27 | 0.10 | 12.9 | 19.2 |
| | Silt | 0.08 | 0.18 | 11.2 | 15.2 |
| | Clay | 0.30 | 0.06 | 10.8 | 12.4 |

^aNo K-fertilization.

^b124 kg K/ha /year.

TABLE VI. DISTRIBUTION OF ¹³⁷CS ACTIVITY WITH DEPTH DOWN TO 25 cm IN ONE PEAT AND THREE MINERAL SOILS 6-8 YEARS AFTER THE CHERNOBYL ACCIDENT OF 1986

| Depth soil 1994 | Peat soil 1992 | Sandy soil 1994 | Silty soil 1994 | Clay |
|-----------------|------------------|-----------------|-----------------|------|
| (cm) | (percent per cm) | | | |
| 0-1 | 6.5 | 28.9 | 11.0 | 17.2 |
| 1-2 | 14.5 | 38.8 | 12.2 | 25.9 |
| 2-3 | 16.1 | 15.0 | 22.9 | 21.2 |
| 3-4 | 16.7 | 6.9 | 15.1 | 19.3 |
| 4-5 | 15.0 | 3.5 | 13.3 | 7.9 |
| 5-7.5 | 2.4 | 1.5 | 5.3 | 1.7 |
| 7.5-10 | 3.1 | 0.2 | 1.9 | 0.3 |
| 10-15 | 1.7 | 0.4 | 0.9 | 0.4 |
| 15-20 | 1.3 | 0.1 | 0.4 | 0.1 |
| 20-25 | 0.4 | - | 0.2 | 0.1 |

6. USE OF ¹³⁷Cs FOR SOIL-EROSION MEASUREMENT

Soil erosion is a global problem [14] and a serious threat to sustainable agriculture. There is less erosion with annual-crop rotations than with cereal monocultures. In Sweden it has been comparatively unimportant, due to its soils and climate, however, counter-measures are required on

some silty and sandy soils, from which considerable erosion has occurred, contributing to nutrient loading of aquatic systems. In fact, much of the work in Sweden on erosion has been related to water quality. Recent results clearly show that soil particle erosion is of importance for the transfer of phosphorus to aquatic systems. Therefore new studies on erosion of agricultural land under Swedish conditions are necessary, and are presently being initiated by the Department of Soil Sciences. The utilization of the Chernobyl and bomb fallout in this research is being developed in a joint project with the Department of Radioecology.

Some areas with silty and sandy soils were rather heavily contaminated by fallout from the Chernobyl accident [10], making them the logical choice as sites for research. As a first step in the application of the ^{137}Cs -technique for soil erosion studies, a site with high Chernobyl fallout of ^{137}Cs has been chosen, just 70 km from our University in Uppsala. The first two objectives are as follows.

1. To investigate sampling methods required to determine the activity of ^{137}Cs per unit area in a silty soil, heterogeneously or homogeneously mixed within the plough layer.
2. To determine the activity of ^{137}Cs in the soil in various parts of cultivated fields to evaluate patterns of erosion and accumulation of soil since 1986.

Step 1 will be carried out in a uniform and levelled field not exposed to erosion in order to develop sampling techniques and investigate small-scale spatial variation in ^{137}Cs activity.

Step 2 will then be carried out in the same field, using a similar sampling technique, in a sloping area that is exposed to erosion, to investigate the extent of redistribution of ^{137}Cs activity.

The Department of Radioecology is in possession of an Ortec computer-aided germanium detector system, housed in a specially designed low-background laboratory. The facility has been used extensively since the Chernobyl accident for measurement of radiocaesium (^{137}Cs , ^{134}Cs) activities in various soil, plant and animal samples. In the proposed research, 330-mL plastic beakers will be filled with soil samples and measured for their activity of both ^{137}Cs and ^{134}Cs .

With experience gained from the pilot experiments, large-scale investigations of soil erosion, using the new ^{137}Cs -technique, are proposed for the west and south of Sweden. Since mainly bomb-source- ^{137}Cs is to be found there, ^{137}Cs -activity measurements will require larger soil volumes.

7. A SPECIAL FRAME TECHNIQUE FOR SOIL SAMPLING

The Department of Soil Science has long and extensive experience in sampling for the determination of depth, soil mass and content of various components in the tilled surface layer of cultivated soils. This experience will be valuable for calibration and interpretation of ^{137}Cs -data.

Sampling will be done from the soil surface to a depth of a few cm below the plough layer, using a frame-scraper sampling technique [15, 16]. A steel frame, 0.5 m^2 in area, is driven into the soil. All soil within this area is sampled and weighed. Complementary core samples are taken to sufficient depth to include all ^{137}Cs present in transects of the slope in order to cover a larger area of the field. After mixing, subsamples are removed for determination of ^{137}Cs activity per unit weight. This allows a good estimation of the activity per unit area. Sampling is done in $3 \times 3 = 9$ points in a grid with a distance of 15-20 m between sampling points. On the slope, complementary points in a grid will be sampled down to the depth of the radionuclide. The pattern of soil erosion and accumulation will be evaluated according to [17].

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USE OF ^{137}Cs AND OTHER FALLOUT RADIONUCLIDES IN SOIL EROSION INVESTIGATIONS: PROGRESS, PROBLEMS AND PROSPECTS

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Abstract

USE OF ^{137}Cs AND OTHER FALLOUT RADIONUCLIDES IN SOIL-EROSION INVESTIGATIONS: PROGRESS, PROBLEMS AND PROSPECTS

Accelerated erosion and soil degradation currently represent serious problems for the global environment. Against this background there is a need to assemble reliable information on the rates of soil loss involved. Existing techniques for documenting rates of soil loss possess many limitations and there is increasing interest in the potential for using fallout radionuclides, particularly ^{137}Cs , to obtain such information. An example of the application of the ^{137}Cs approach to a cultivated field at Rufford Forest Farm, Nottinghamshire, UK, is presented to illustrate its value. The key advantages of the approach are that it provides a means of assembling retrospective estimates of medium-term (ca. 40 years) rates of soil loss and the spatial pattern of erosion and deposition involved, on the basis of a single site visit. There are, however, currently a number of problems and uncertainties associated with the use of ^{137}Cs in soil erosion investigations, and these are reviewed and needs for further research identified. Potential developments of the approach, including the use of other fallout radionuclides such as unsupported ^{210}Pb and ^7Be are also considered.

1. THE CONTEXT

Although much of the recent concern for the global environment has focussed on problems of global warming and climatic change, there is also growing evidence that accelerated erosion and associated soil degradation represent a major problem for the sustainable development of agricultural production in a world characterized by a rapidly expanding population. Recent assessments of the global soil-erosion problem afford considerable cause for concern. For example:

- Brown [1] reports that each year the world is currently losing 23×10^9 tonnes of soil from croplands in excess of new soil formation. This is equivalent to a depletion of the global soil resource by 7% each decade.
- ISRIC [2] indicates that the soils of an area covering more than 8% of the land surface of the globe (i.e. $> 10^9$ ha) have now been degraded by water erosion.
- Buringh [3] estimates that the global loss of agricultural land due to soil erosion is now of the order of 3×10^6 ha per year.
- Pimental et al. [4] report that of the 11.6×10^6 ha of forest cleared annually, more than half can be attributed to agricultural soil degradation and subsequent expansion of agriculture onto new land.
- Brown and Young [5] estimate that current rates of soil loss from croplands result in an annual loss to the global grain output of 9×10^6 tonnes.

To date, much of the loss of soil and agricultural land identified above has been compensated by clearing of new land for crop production and by use of fertiliser and improved crop strains to increase yields on existing land, but the scope for maintaining such compensation measures will clearly

decline in the future. In addition to *on-site* costs associated with reduced soil productivity and loss of agricultural land, there is also increasing evidence that the *off-site* costs of soil erosion related, for instance, to increased sediment transport in rivers, may be of equal, if not greater, importance. Mahmood [6], for example, has reported that the world's reservoirs are currently losing storage at a rate of about 50 km³ per year as a result of sedimentation. This is equivalent to a storage loss of approximately 1 % per year. Taking account of the central importance of reservoirs for domestic and industrial water supply, for irrigation schemes, and for hydropower production, and thus for economic progress in many developing countries, such losses of storage are of considerable concern. Annual replacement costs are conservatively estimated at about US\$6 x 10⁹ per year and the new reservoir sites needed to replace lost storage are increasingly difficult to find.

Against this background, there is an increasing need to assemble reliable information on rates of soil erosion or soil loss in different areas of the world. Such information is, for example, needed to assess the magnitude of the problem, to evaluate the key factors influencing rates of soil loss and to investigate erosion-crop productivity relationships. This need is, however, not readily met by existing methods of measuring soil erosion, such as erosion plots, which possess many important limitations in terms of cost, representativeness and the reliability of the resulting data [7, 8]. These methods are also generally unable to provide the detailed spatially distributed data required to verify the new generation of distributed erosion and sediment yield models and to interface with current developments in the application of GIS and geostatistics to this field. Recent work in exploring and exploiting the possibilities for using fallout radionuclides, and more particularly ¹³⁷Cs, to document rates and patterns of soil redistribution can, however, be seen as overcoming many of the limitations associated with existing techniques for monitoring soil erosion and as offering considerable potential for meeting the needs outlined above.

2. THE USE OF ¹³⁷Cs MEASUREMENTS

2.1. The basis

The potential for using ¹³⁷Cs measurements to investigate rates and patterns of soil loss was originally recognised by Ritchie and McHenry in the USA [9] and the basis of the approach is now well documented [10, 11, 12, 13, 14]. In essence, the approach is founded on the fact that radiocaesium released into the stratosphere as a by-product of past atmospheric testing of thermonuclear weapons during the 1950s and early 1960s reached the land surface as fallout and was in most environments rapidly and strongly adsorbed by the surface soil. Its subsequent redistribution can be attributed to the erosion, transport and deposition of soil particles and measurements of the current distribution of ¹³⁷Cs within the landscape provide a means of establishing rates of erosion and deposition, and the spatial patterns involved, during the period since the main phase of atmospheric fallout. Assessment of ¹³⁷Cs redistribution is commonly based on comparison of the measured inventories (total activity per unit area) at individual sampling points, with an equivalent estimate of the inventory representing the cumulative atmospheric fallout at the site, taking due account of the different behaviour in cultivated and non-cultivated soils. Because direct long-term measurements of atmospheric fallout are rarely available, the cumulative input or reference inventory is usually established by sampling adjacent undisturbed, uneroded locations, generally under permanent pasture, which can provide an estimate of total fallout inputs. The magnitude and direction of measured deviations from the local reference level provide a qualitative assessment of sediment redistribution.

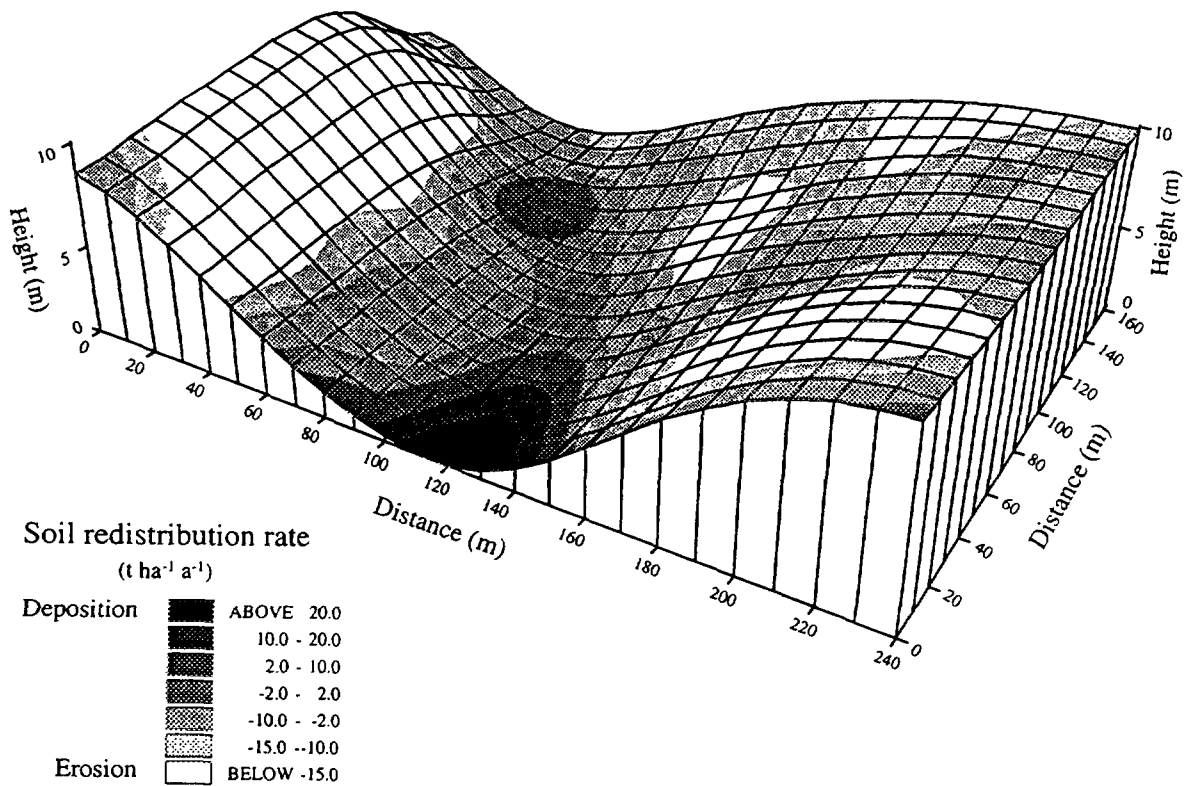


FIG. 1. Medium-term (35 years) erosion and deposition rates within a field at Rufford Forest Farm, Nottinghamshire, UK, estimated using ¹³⁷Cs measurements.

To derive quantitative estimates of the rates of erosion and aggradation involved, it is necessary to establish a relationship between the magnitude of the deviation from the reference inventory and the extent of soil loss or gain [15]. Because empirical calibration data are rarely available, many workers have favoured the use of theoretical relationships or models to provide the necessary calibration function. Such models can be used to simulate the effect of a range of long term erosion and aggradation rates upon the ¹³⁷Cs inventory of soil profiles, and the resultant data can be used to derive the calibration relationship [15, 16].

2.2. An example

The advantages and potential value of the ¹³⁷Cs approach to documenting medium-term (i.e. ca. 40 years) rates and patterns of soil erosion can be usefully demonstrated by introducing an example from a study of a 3.8 ha cultivated field at Rufford Forest Farm, Nottinghamshire, UK, reported by Walling & Quine [10]. This area, which is underlain by brown sand soils of the Cuckney 1 association, is primarily used for arable cultivation. Sugar beet is widely grown and soil erosion has frequently been observed in the area [17]. A 20m x 20m grid was used as a basis for collecting soil cores from the field and a total of 117 cores was obtained using a motorized percussion corer (38 cm²) inserted to a depth of 60 cm. The reference cores were obtained from an area of uneroded, undisturbed grassland located 1.7 km from the study field. After collection, all cores were air dried and lightly ground and the ¹³⁷Cs content of the material passing through a 2 mm sieve was determined by gamma spectrometry using an HPGe coaxial detector. A value of 3200 Bq m⁻² was obtained for

TABLE I. SPATIALLY INTEGRATED ESTIMATES
OF SOIL REDISTRIBUTION IN THE FIELD AT
RUFFORD FOREST FARM ILLUSTRATED IN FIG. 1.

| Component | Estimate |
|---|----------|
| Gross erosion rate (t ha ⁻¹ yr ⁻¹) | 12.2 |
| Eroding zone | |
| Mean erosion rate (t ha ⁻¹ yr ⁻¹) | 13.8 |
| Fraction of total area (%) | 89 |
| Fraction of total area with erosion rates | |
| > 2 t ha ⁻¹ yr ⁻¹ | 81 |
| > 4 t ha ⁻¹ yr ⁻¹ | 72 |
| Aggrading zone | |
| Mean aggradat'n rate (t ha ⁻¹ yr ⁻¹) | 16.1 |
| Fraction of total area (%) | 11 |
| Net erosion rate (t ha ⁻¹ yr ⁻¹) | 10.5 |
| Sediment delivery ratio (%) | 86 |

TABLE II. SOME ADVANTAGES OF THE ¹³⁷Cs TECHNIQUE FOR ESTIMATING RATES OF
SOIL LOSS AND DEPOSITION

- (1) Estimates relate to individual points within the landscape and information relating to both rates and spatial patterns of soil redistribution can be assembled
- (2) The technique is capable of providing spatially-distributed data which are compatible with recent advances in physically-based distributed modelling
- (3) The estimated rates of soil redistribution reflect the integration of all landscape processes (e.g. water and wind erosion, tillage effects etc.)
- (4) Estimated rates of soil redistribution relate to the past 40 years and thus provide estimates of longer-term average rates of erosion and deposition. Short-term measurements may be unrepresentative
- (5) There are no major scale constraints apart from the number of samples that can be processed. Areas studied can range from a few m² to small drainage basins (e.g. 5 ha)
- (6) Application of the technique does not involve major disturbance of the landscape under study
- (7) Estimates can be obtained on the basis of a single site visit
- (8) Estimates based on contemporary sampling are retrospective and therefore avoid the need for establishment of long-term monitoring programmes

the local reference inventory. Deviations of the inventories associated with the individual sampling points within the field, from the reference value, were calculated, and a calibration model was used to derive estimates of the rates of erosion and deposition occurring within the field over the past 35 years. The resultant pattern of soil redistribution documented for the field, which is presented in Fig. 1, reflects the action of a range of erosion processes, including both water erosion and the effects of soil tillage.

In addition to providing an assessment of the spatial pattern of soil redistribution within the study field, the individual point estimates of erosion and deposition rates derived from the ^{137}Cs measurements can be spatially integrated to produce a range of measures of the overall status of erosion and deposition within the field (Table I). The values of both the gross erosion rate and the mean erosion rate for the eroding sites permit clear assessment of the severity and potential on-site impact of erosion within the area under investigation. Furthermore, assessment of both the net soil loss and the sediment delivery ratio permits an evaluation of the potential for off-site problems posed by sediment leaving the field and entering local watercourses.

The example outlined above is based on measurements of the total inventory of individual soil cores, since this minimises the number of samples requiring gamma assay and therefore reduces the analytical demands. Further development and refinement of the approach could, nevertheless, usefully involve consideration of the depth distribution of radiocaesium in the soil profile, since such information can afford the basis for improved interpretation of the erosional history of the study site, particularly in areas of intensive land use [18].

The information generated for the field at Rufford Forest Farm from ^{137}Cs measurements, which is presented in Fig. 1 and Table I represents essentially unique data that would be effectively impossible to assemble by any other means. Use of erosion plots would, for example, provide information only on the net soil flux at the lower end of the bounded plot and would be unable to document the *spatial pattern* of erosion and deposition within the field. Equally, they would not document the effects of soil redistribution by tillage. Furthermore, although the medium-term and temporally-lumped nature of the erosion rate estimates provided by the ^{137}Cs measurements could be seen as a limitation, it must be recognised that short-term measurements could be unrepresentative and that long-term monitoring is likely to prove both costly and labour intensive and in many instances impractical. In addition, the ^{137}Cs approach affords a unique opportunity to secure *retrospective* information for a particular site and to acquire the *spatially distributed* data required to develop and validate the new generation of distributed soil-loss models currently being developed [19]. These and other key advantages of the use of ^{137}Cs to monitor rates and patterns of soil loss are summarized in Table II.

3. CURRENT PROGRESS

Although the potential for using ^{137}Cs as a tracer in soil erosion investigations was recognised at a relatively early stage in the United States of America [9, 20], wider application of the approach was relatively slow. It is only in more recent years that its potential has been more generally exploited. Figure 2 provides a schematic representation of the expansion of such exploitation over the past 30 years by plotting the activity of individual research groups who have made extensive use of the approach. Figure 2 is neither comprehensive nor precise, since some groups will have been excluded and the periods of activity may be incorrect. Nevertheless, it clearly demonstrates the very significant expansion in the application of the approach in recent years. Such applications have

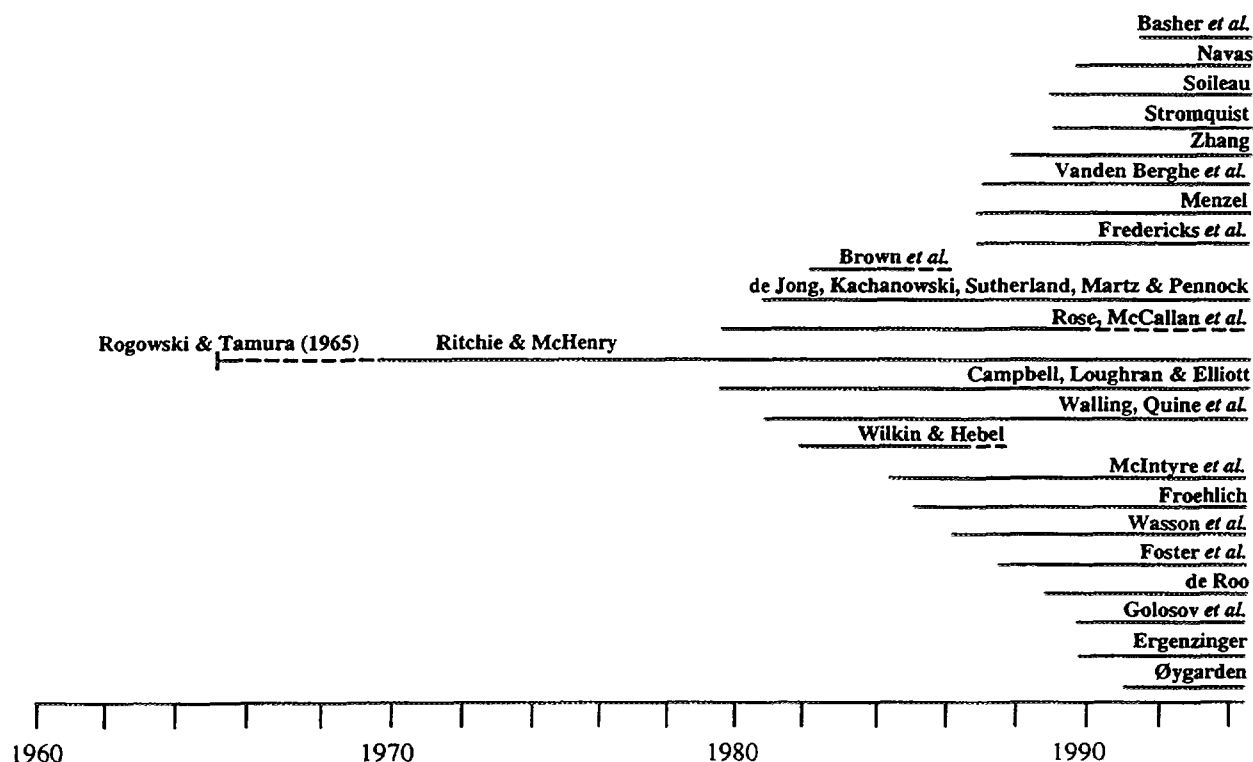


FIG. 2. The evolution of the use of ^{137}Cs in soil-erosion studies.

encompassed a wide variety of locations throughout the world (Fig. 3), ranging from glacierized mountain areas in Greenland [21] and mountain areas in Sweden [22], through the prairie and steppe regions of Canada and the Russian Federation [23, 24, 25, 26] and semi-arid areas of Spain [27], to tropical areas of Africa [28] and Thailand [29]. Therefore, the approach must now be seen as having global relevance [30, 31].

4. PROBLEMS AND UNCERTAINTIES

Although the use of ^{137}Cs measurements to document rates and patterns of soil erosion must now be seen as well established and proven, and as providing a valuable alternative to traditional approaches, a number of limitations and uncertainties must be recognised. These are briefly reviewed.

4.1. Global and regional patterns of ^{137}Cs fallout

Use of ^{137}Cs measurements to document rates and patterns of erosion requires that existing inventories should be sufficiently high to permit accurate assessment of radiocaesium redistribution during the period since the main period of weapons testing. Measurement precision is particularly important in this context, since it is difficult to obtain reliable measurements of ^{137}Cs activity when inventories are low. Although a detailed assessment of the global pattern of ^{137}Cs inventories in undisturbed soils, which would be representative of fallout inputs, has yet to be undertaken, it is

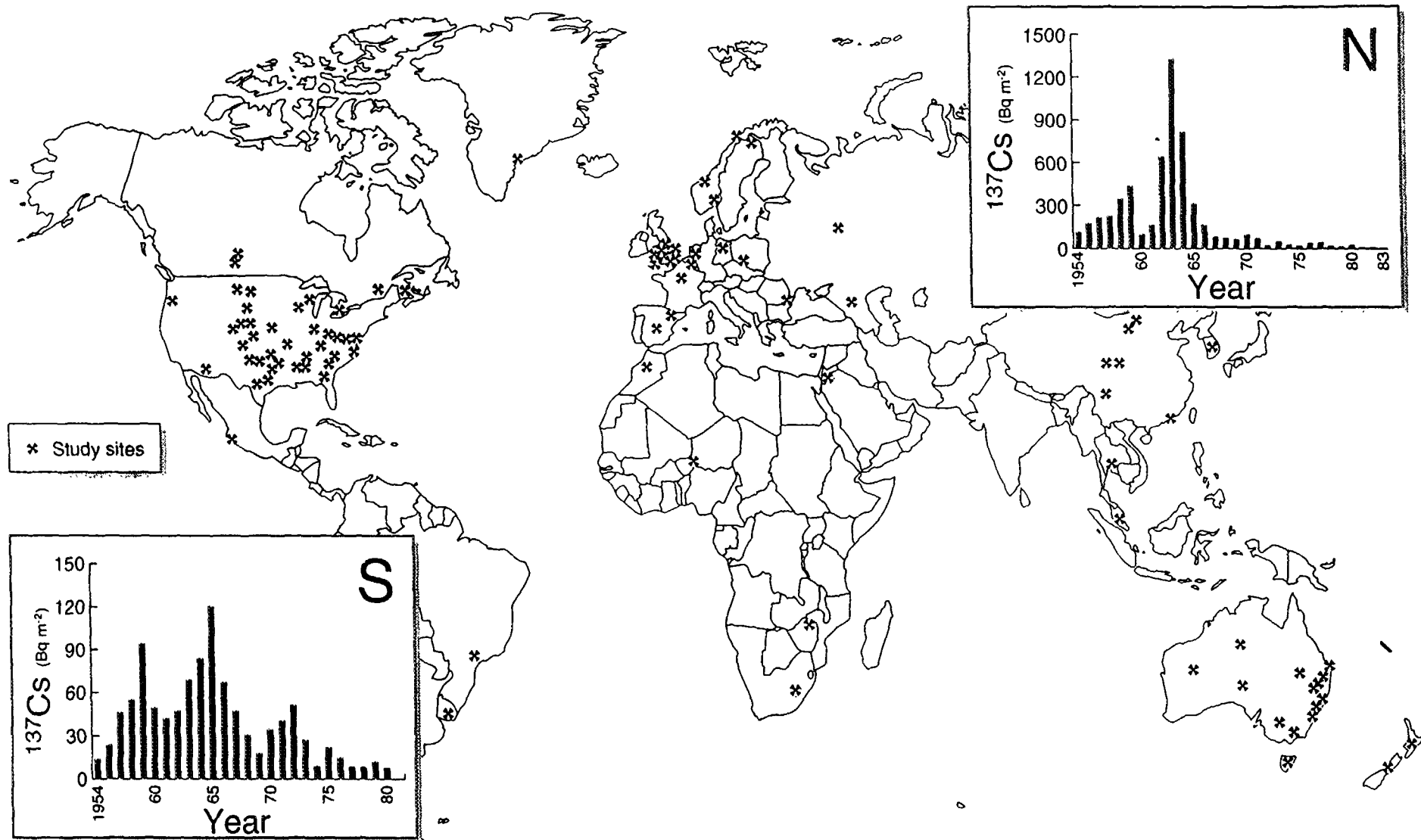


FIG. 3. Locations where ^{137}Cs has been used successfully in soil-erosion and related studies and typical fallout records for the northern (New York, USA/Milford Haven, UK) and southern (Adelaide/Brisbane) Australia) hemispheres.

known that such inventories are much lower in the southern hemisphere than in the northern hemisphere (cf. Fig. 3) and that inputs to equatorial areas were probably considerably lower than those in the mid-latitude areas of Europe and North America, where most work on exploiting the potential of ^{137}Cs measurements has been undertaken. Table III presents generalised information on the global distribution of the fallout of ^{90}Sr , another fission product of weapons testing, based on data reported by Larsen [32]. In view of the common origin of both strontium-90 and caesium-137, the global pattern of fallout of both radionuclides can be expected to be similar.

Table III indicates that in equatorial areas and over large regions of the southern hemisphere, ^{137}Cs inventories are likely to be <30% of those recorded in the mid-latitudes of the northern hemisphere. There have, nevertheless, been many reports of the successful application of ^{137}Cs measurements in areas of the southern hemisphere with relatively low inventories such as Australia and southern Africa [33, 34], but uncertainty remains over the viability of the approach in arid regions and some equatorial areas.

4.2. Chernobyl fallout: problems and opportunities

The use of ^{137}Cs measurements for soil-erosion assessment is generally based on the assumption that it is possible to assess the redistribution of radiocaesium occurring during the period extending from the main period of weapons test fallout in the late 1950s and the 1960s to the present, and to use this information to estimate rates of erosion and deposition. This basis will be compromised in areas that received significant amounts of Chernobyl-derived ^{137}Cs fallout in 1986. Such areas include large parts of northern, eastern and western Europe. In the first place, it will no longer be possible to assume that the documented redistribution reflects the net effect of soil erosion processes operating over a period of ca. 40 years. Some of the ^{137}Cs inventory will have been present in the soil only since 1986. Although it was possible to distinguish the bomb- and Chernobyl-derived components of the total ^{137}Cs inventory at the time of the Chernobyl incident by measuring the ^{134}Cs activity (^{134}Cs was associated only with Chernobyl fallout and exhibited a fixed ratio to ^{137}Cs), this is no longer possible due to the short half life of ^{137}Cs (2.2 years) which is now below the level of detection at most locations. Secondly, the main input of Chernobyl fallout was restricted to a short period immediately after the accident and was in many places characterised by marked spatial variability due to the interaction of the plume with local air mass and precipitation dynamics [35]. This variability introduces problems in terms of establishing a local reference level against which the inventories for individual cores can be compared in order to assess the gain or loss of radiocaesium and thus the rate of soil redistribution.

Although the addition of Chernobyl fallout will inevitably complicate the interpretation of ^{137}Cs measurements in many areas, some potential undoubtedly exists to make use of the two phases of fallout input to derive additional information on the erosional history of a site. Chernobyl inputs could be incorporated into theoretical calibration procedures aimed at interpreting the inventory values for bulk cores, but in most cases it may prove more useful to consider the depth distribution of radiocaesium within the soil, which may in turn reflect both input phases. Where Chernobyl-derived ^{137}Cs inventories are very much greater than bomb-derived inventories, such that the latter may be effectively ignored, it should prove possible to use the redistribution of Chernobyl-derived ^{137}Cs to assess soil redistribution during the period since 1986.

TABLE III. LATITUDINAL VARIATION OF BOMB-DERIVED ^{90}Sr INVENTORIES AT THE END OF 1983, BASED ON LARSEN [32]

| Latitude band | Mean ^{90}Sr inventory | |
|------------------|------------------------------------|------------------------|
| | Northern hemisphere | Southern hemisphere |
| | (Bq m ⁻²) ^a | |
| 0-10 | 953 | 526 |
| 10-20 | 1370 | 491 |
| 20-30 | 2075 | 841 |
| 30-40 | 2862 | 967 |
| 40-50 | 3867 | 1124 |
| 50-60 | 3585 | 672 |
| 60-70 | 2084 | 455 |
| 70-80 | 897 | 276 |
| 80-90 | 409 | 154 |

^a Inventory values have been estimated from the cumulative ^{90}Sr deposition on the land and ocean surfaces of individual latitudinal belts.

4.3. Local variability of ^{137}Cs fallout and establishment of local reference inventories

As indicated above, most procedures involving the use of ^{137}Cs measurements to estimate rates of soil redistribution are based on a comparison of measured inventories for individual soil cores with a local reference inventory. In most investigations it is assumed that this local reference inventory will be effectively constant over a small study area. This assumption can be readily justified theoretically in terms of the extended period of input associated with bomb fallout, such that local variability associated with individual precipitation events is likely to disappear when the inputs associated with a large number of precipitation events are averaged. However, more attention undoubtedly needs to be given to justifying this assumption empirically and assessing the potential magnitude of local variability in fallout inventories. This in turn requires consideration of potential sources of sampling variability, in order to distinguish true spatial variability from variability introduced by measurement precision and sample collection [36, 37, 38]. The latter could be seen as effectively adding confidence limits to any estimate of the local reference inventory, such that this should be represented by a range rather than an absolute value. Equally, where there is clear evidence of true spatial variability of the local reference inventory, it will be necessary to consider whether this is essentially random or if there is a systematic trend. In the former case the confidence limits associated with the estimate of the local reference inventory will need to be increased, whereas in the latter case appropriate spatial statistics may be used to represent the pattern or trend involved. Where random spatial variability exists, it is

important to consider the number of samples required to obtain a reliable estimate of the mean and variance of the reference inventory at the required level of confidence. Results presented by Owens and Walling [36] indicate that confidence limits associated with measurement precision and sampling variability are likely to be of the order of $\pm 10\%$ and $\pm 5\%$ respectively at the 95% level of confidence.

4.4. Behaviour of ^{137}Cs within the soil profile

Use of ^{137}Cs measurements to estimate rates of soil redistribution depends heavily upon the Assumption that radiocaesium fallout is rapidly and strongly adsorbed by the upper horizons of the soil and that its subsequent redistribution therefore reflects movement of soil particles. There have been a number of reports of the mobility of Chernobyl-derived radiocaesium in the receiving soils that could cause this assumption to be questioned. However, it is important to recognise that most of these reports relate to upland areas with acid, highly organic soils with limited capacity to adsorb radiocaesium. Furthermore, the concentrations of radiocaesium in precipitation associated with Chernobyl fallout were frequently orders of magnitude greater than those associated with bomb fallout, which occurred over a period of many years rather than few days, and may therefore not provide a meaningful replication of the response of soils to bomb fallout.

Most agricultural soils, which will be investigated in soil erosion studies, can be expected to conform to the assumptions of rapid and strong fixation of radiocaesium associated with bomb fallout and there have been many laboratory and field investigations that have confirmed such behaviour. Livens and Loveland [39], for example, cite the work of several investigators as demonstrating the highly efficient extraction of radiocaesium from dilute (0.001M) solutions by clay minerals. The radiocaesium concentrations in the solutions are several orders of magnitude greater than those associated with rainfall during the period of weapons testing fallout. The effects of soil texture and the magnitude of the clay fraction must also be considered, but other studies indicate that the proportions of fine particles commonly found in mineral soils do not limit radiocaesium adsorption. Livens and Baxter [40] examined a range of soil types and found that radiocaesium has been adsorbed by all the mineral soils investigated. Strong adsorption is also reflected by the low rates of vertical migration of ^{137}Cs evident for many soil types in both field and laboratory experiments [41, 42] and in the depth distributions of the weapons testing ^{137}Cs characteristic of undisturbed soil profiles.

Figure 4 illustrates typical ^{137}Cs depth distributions for a selection of soils investigated by the author. These encompass a textural range from clay to sand in the UK (Figs. 4(a) to (e)) and an environmental range from temperate through semi-arid to subtropical, worldwide. All exhibit a sharp decline in ^{137}Cs activity with increasing depth and in all cases more than 75% of the total inventory is found in the top 15 cm, indicating that downward translocation is minimal. Furthermore, the total inventories of the UK soils are in close agreement with existing evidence regarding total fallout amounts [43]. These profile characteristics again support the assumption that in most environments the majority of mineral soils have the capacity to adsorb and immobilize fallout. Further work is, however, required to investigate the behaviour of highly weathered tropical soils with respect to the fate of radiocaesium fallout inputs.

4.5. Preferential mobility of ^{137}Cs in response to grain size effects and the organic fraction

When using ^{137}Cs measurements to estimate rates of soil erosion or deposition, it is frequently assumed that there is a simple relationship between the percentage increase or decrease of the radiocaesium inventory relative to the local reference value and the mass of soil eroded or deposited. Thus, in the case of the *proportional method*, which is frequently used to convert measurements of the percentage reduction in the total ^{137}Cs inventory to an estimate of the erosion rate, the soil loss is assumed to be directly proportional to the amount of ^{137}Cs removed from the plough layer [15]. A 50% reduction in the ^{137}Cs inventory would be taken as indicating that a depth of soil equivalent to 50% of the depth of the plough layer had been eroded during the period since the commencement of fallout. The assumption of proportionality is, however, an oversimplification of reality, since the surface lowering by erosion may be associated with preferential mobilisation of specific size fractions or the organic fraction. It is well known that the radiocaesium fixed within a soil will be preferentially associated with the finer fractions, and if there is selective erosion of these finer fractions the proportional method will overestimate the rate of surface lowering. Equally, if the coarser fractions

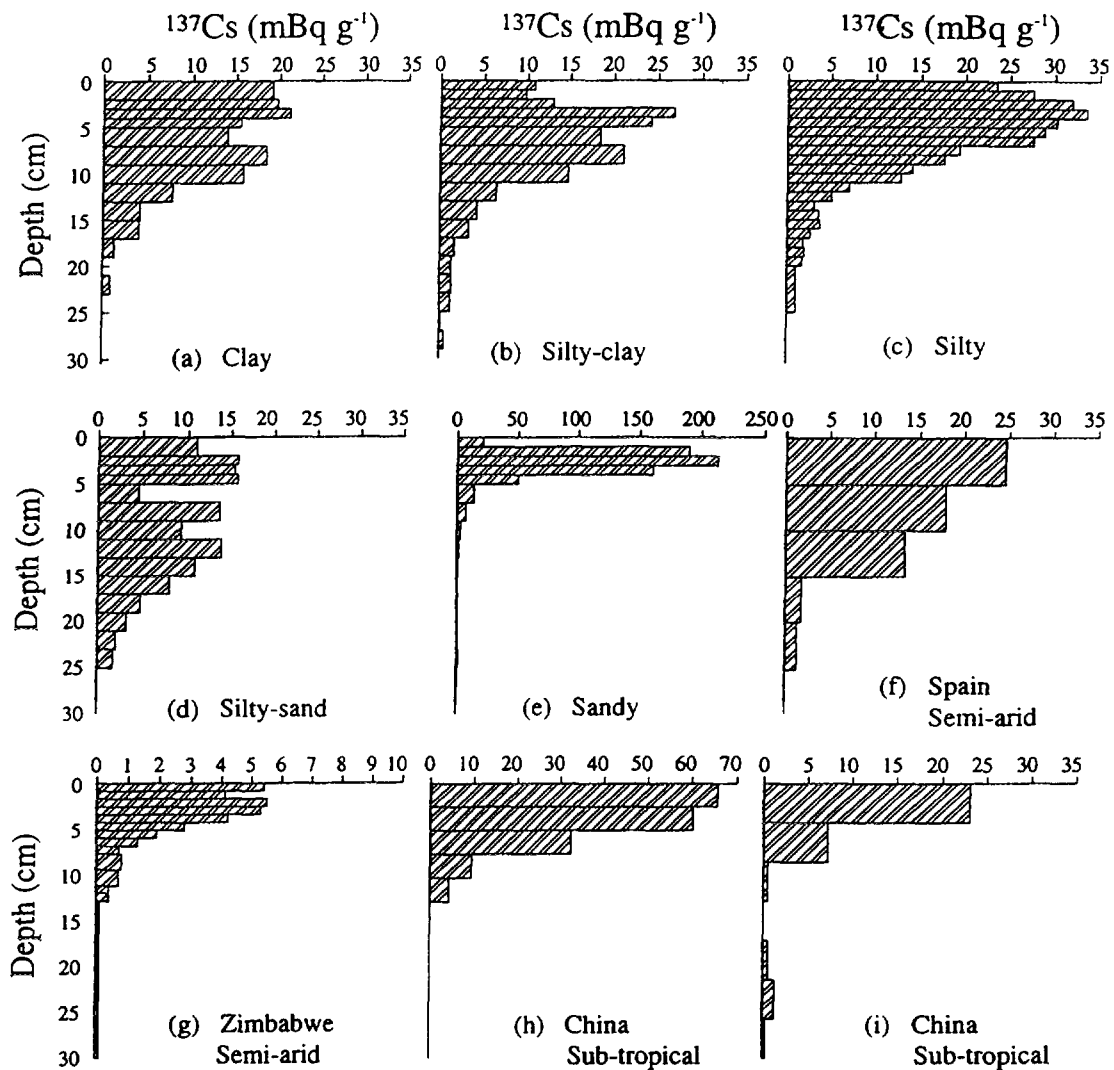


FIG. 4. Typical ^{137}Cs profiles associated with undisturbed soils: (a) to (e): soil-textural variation in the UK; (f) to (i): environmental variation worldwide.

are preferentially mobilised, the rate of surface lowering is likely to be underestimated. Similarly, depending on whether the organic fraction is enriched or depleted in radiocaesium relative to the mineral soils, preferential removal of this fraction would again cause the erosion rate to be over- or underestimated respectively. In many situations, material mobilized by erosion will be eroded in the form of aggregates containing primary particles of a range of sizes and both mineral and organic particles [44]. Under such circumstances, the potential for selective erosion is likely to be limited. Nevertheless, there is a need for increased attention to this facet of the behaviour of radiocaesium in eroded soil and for incorporation of the enrichment ratio concept [45], which has been extensively used in studies of sediment-associated contaminant transport, into the interpretation of ^{137}Cs measurements.

The implications of selective mobilisation and transport are probably more significant in the use of ^{137}Cs measurements to estimate rates of deposition or aggradation, since selective deposition of coarser particles will frequently occur. If, as is likely, the resulting deposits are characterised by lower radiocaesium concentrations than the parent soil, application of simple proportionality assumptions will lead to underestimation of the deposition rates involved. An improved understanding and representation of the effects of particle size in the adsorption of radiocaesium, such as attempted by He and Walling [46], is required to provide a basis for improved interpretation of ^{137}Cs measurements in soil erosion investigations.

4.6. Establishing improved 'calibration' relationships between measured inventories and rates of soil loss or deposition

Although ^{137}Cs measurements can readily provide *qualitative* information concerning the redistribution of soil within a field or landscape, there will be a requirement in most studies to convert such measurements into *quantitative* estimates of erosion and deposition rates. This will commonly involve application of a 'calibration' relationship that relates the percentage increase or decrease of the radiocaesium inventory at the measuring point, relative to the local reference value, to the rate of erosion or deposition involved. Walling and Quine [15] have reviewed many of the uncertainties and inconsistencies associated with the wide range of calibration relationships employed in existing investigations, and the scale of the problem is clearly demonstrated in Fig. 5, which is based on their work and which plots a number of these calibration functions on common coordinates.

Estimates of the erosion rate associated with a particular level of ^{137}Cs can vary by more than an order of magnitude, according to the calibration relationship used. Some of this variability is undoubtedly related to local conditions, since variations in cultivation practices, and particularly plough depths, will be reflected in different relationships. Furthermore, for a given rate of soil loss, the reduction in the ^{137}Cs inventory might be expected to vary according to the relative importance of sheet and rill erosion. In some cases relationships have been incorrectly derived, but much of the variability evident in Fig. 5 is a reflection of the uncertainty surrounding the precise nature of the calibration relationship and the major controlling factors.

Existing calibration relationships essentially fall into two categories. The first involves empirical relationships between measured or estimated soil loss or deposition and measurements of ^{137}Cs depletion or gain. The second employs theoretical models or accounting procedures to derive the relationship. Both categories possess limitations. In the first case, data suitable for deriving empirical relationships are very limited for most areas of the world. In the second, there is a need to represent accurately the long-term behaviour and fate of radiocaesium fallout in the soil of the study area. The inadequacy of the simple proportional model in terms of representing the grain size selectivity of

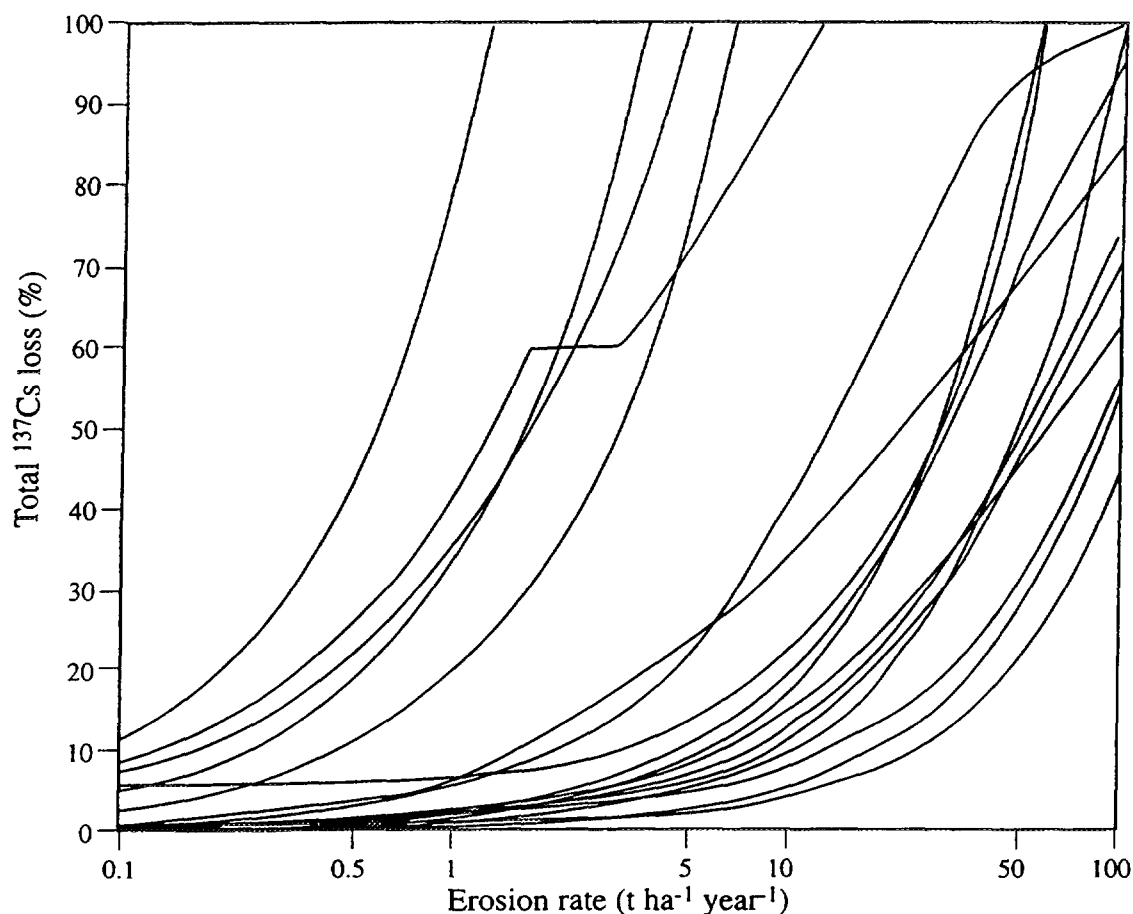


FIG. 5. A selection from the range of calibration relationships between the percentage of the total ^{137}Cs inventory lost from a soil profile and the long-term erosion rate used by various workers. (based on Walling and Quine [15]).

erosion, transport and deposition processes has already been highlighted above, but these and other considerations can be readily incorporated into more comprehensive theoretical models and accounting procedures [15, 16].

Much of the uncertainty and inconsistency surrounding calibration relationships evident in Fig. 5 has been reconciled by the development of improved theoretical models and accounting procedures [15, 16] and by the careful assessment of data obtained from erosion plots [47, 48]. Provided that care and critical appraisal are exercised, calibration problems should not be seen as a major impediment to the wider application of ^{137}Cs measurements in soil-erosion investigations. The author has favoured the application of theoretical accounting procedures that are able to represent the combined effect of all redistribution processes operating over the period since the initiation of atmospheric fallout and to take account of the known history of land management at the site, to establish site-specific calibration relationships [15, 34]. Where independent evidence of longer-term erosion rates exists, use of these procedures has resulted in close agreement of the estimates produced [49]. Nevertheless, more work is undoubtedly required to make use of available long-term erosion plot experiments in validating and developing empirical calibration relationships, to refine theoretical procedures, for example by incorporating the particle-size selectivity of erosion and deposition processes and by experimental investigations [e.g. 50, 51] and to combine these two approaches.

4.7. Past fallout behaviour

When interpreting ^{137}Cs measurements in terms of soil-redistribution processes and when attempting to develop comprehensive representations of the fate of radiocaesium in agricultural soils for use in developing theoretical accounting models, one of the greatest sources of uncertainty relates to the precise fate and behaviour of bomb-derived radiocaesium at the time of deposition during the late 1950s and early 1960s. For example, it is important to know the likely thickness of the surface layer of soil enriched with radiocaesium fallout. If this was extremely thin, a small amount of surface erosion could remove the recently deposited ^{137}Cs , whereas if the thickness was greater a substantial proportion of the recent input would remain for subsequent incorporation into the soil by tillage activity. Equally, it is important to relate the pattern of fallout receipt during the year to the precise timing of tillage and erosion events, since the proportion of the recent fallout removed by erosion will depend on whether it still remains at the surface or is incorporated into the plough layer. It is impossible to recreate the conditions of the late 1950s and 1960s, but additional evidence of the likely behaviour and fate of contemporary fallout could be obtained from carefully designed experiments and from collation of available records of the magnitude and timing of fallout inputs.

4.8. Physiographic constraints on the application of ^{137}Cs measurements in soil erosion investigations

It is important to recognise that ^{137}Cs measurements are primarily applicable to investigations of sheet and rill erosion, which involve gradual surface lowering. In the case of rill erosion, subsequent tillage will redistribute and smooth the surface soil such that its net effect in terms of the radiocaesium content of the soil is effectively similar to that of sheet erosion. Caesium-137 measurements are of limited value in investigating gully erosion, since the depths of incision are very much greater than the depth of soil labelled with radiocaesium. Nevertheless, ^{137}Cs measurements can be used in gully erosion investigations to assist in establishing the chronology of incision and deposition, since the surface layers of sediment deposited after the late 1960s are likely to contain little or no radiocaesium. Equally, when tracing the source of sediment transported by a river, the ^{137}Cs content of the sediment can provide evidence as to the likely source of the sediment [52, 53]. Where sheet and rill erosion represent the dominant sources, radiocaesium concentrations are likely to be relatively high, whereas if the sediment originates primarily from gully erosion the concentrations are likely to be very low.

Problems of low fallout inventories in some areas of the world highlighted above may also limit the applicability of ^{137}Cs measurements in certain physiographic zones. These are likely to include many of the desert areas of the world. In addition, several assumptions of the approach could be violated in areas where a substantial proportion of the precipitation falls as snow and accumulates through the winter season. The effects of drifting in redistributing snow will also introduce spatial variability of fallout receipt, such that it will be more difficult to establish a local reference inventory and variations in radiocaesium inventories across an area could reflect variations in input rather than erosional redistribution. Accumulation of drifting snow in hollows and its removal from ridge tops could, for example, falsely suggest the existence of depositional areas in the former and eroding areas on the latter.

4.9. Costs and operational constraints and the need for upscaling

The various advantages of the ^{137}Cs approach to soil-erosion assessment listed in Table 2 underpin its increasing use. In particular, the potential for obtaining retrospective measurements of soil erosion and deposition on the basis of a single site visit represents a very important advantage. Similarly, the potential for obtaining detailed distributed data matches the requirements of contemporary developments in the application of distributed erosion and sediment yield models. Nevertheless, there are significant cost and operational constraints on its use. More particularly, the gamma spectrometry analysis required to establish the ^{137}Cs inventories of bulk cores or the ^{137}Cs content of depth incremental samples from sediment cores are costly in terms of detector time. Because of the relatively low levels of radiocaesium activity involved, count times for individual samples are typically of the order of 10,000 to 30,000 s. This in turn limits the number of samples that can be processed, since equipment costs generally preclude the establishment of more than a small number of detectors. Collection of substantial numbers of bulk cores can also represent a time consuming exercise in areas characterised by dense dry soils, but this limitation can be overcome by use of motorised percussion corers.

Because of the limitations on sample numbers imposed by analytical constraints, and the spatial variability of soil erosion that precludes using a few radiocaesium measurements to derive estimates of soil loss for a large area, most investigations making use of ^{137}Cs measurements have focussed on small areas. For example, in the case of the field at Rufford Forest Farm, Nottinghamshire, UK, studied by Walling and Quine [10] and shown in Fig. 1, a total of 117 bulk cores were collected from the 3.8 ha field using a 20 m x 20 m grid. Similarly, the reconnaissance survey of soil erosion in Australia reported by Loughran et al. [33], which allocated 500 ^{137}Cs analyses per state, focussed on the detailed investigation of a small number of representative slope profiles. There is clearly a need to develop means of scaling up the use of ^{137}Cs measurements to provide data from larger areas. This could be achieved by using the point-specific information provided by ^{137}Cs measurements as 'ground truth' in extrapolation procedures [11]. These procedures could involve the development of GIS erosion models for use with Digital Elevation Models (DEMs), such that the ^{137}Cs measurements could be used to calibrate the model that could then be applied to a wider area using topographic data. Similarly, there is potential to couple the use of ^{137}Cs measurements and satellite imagery, such that the former could provide ground truth for areal extrapolation procedures based on the latter [30, 54].

5. FUTURE PROSPECTS

Although the ^{137}Cs approach has been employed in an increasing number of soil erosion investigations in recent years, considerable scope remains to refine the procedures employed and to exploit its potential more fully. To date, most of the studies employing this approach have been concerned with soil erosion by water, but there is clearly potential to extend its application to redistribution of soil by tillage [55, 56] and to wind erosion [57]. Where it is possible to initiate longer-term investigations, it should also be feasible to extend the approach to include resampling of soil inventories to permit comparison of values for known dates and thus estimate the erosion rates for the associated intervening periods. Care would, however, need to be exercised to ensure that the period between measurements was sufficiently long to produce significant differences in recorded

inventories, bearing in mind the precision errors typically associated with gamma assays. This strategy has the attraction of avoiding the need to establish the reference inventory for a site, but it no longer affords the potential to obtain retrospective estimates of soil redistribution.

The distributed data generated by ^{137}Cs measurements are also ideally suited to coupling with GIS and spatial statistics [58] and for use in verifying distributed erosion and soil loss models [59]. Scope also clearly exists for using estimates of soil loss derived from radiocaesium measurements as a basis for establishing medium-term soil loss / crop productivity relationships. In addition, there is considerable potential for extending the use of ^{137}Cs as a sediment tracer from consideration of soil redistribution within individual fields to investigation of the movement and storage of sediment within a drainage basin. Walling and Bradley [60] have, for example, shown how ^{137}Cs measurements can provide the basis for establishing the sediment budget for a small drainage basin, and the successful application of ^{137}Cs measurements in fingerprinting sediment sources and in estimating rates of overbank floodplain accretion has been demonstrated by a number of studies [52, 61, 62, 63, 64].

A further area to which effort could be profitably directed is exploitation of the potential for using other fallout radionuclides. Caesium-137 has attracted particular attention for application in soil erosion investigations by virtue of its high affinity for sediment particles, its relatively long half-life, its comparative ease of measurement and its well-defined temporal pattern of fallout input. As such it has effectively dominated most, if not nearly all, recent work and there have been very few attempts to use other fallout radionuclides that might offer similar or complementary opportunities. Thus, for example, the possibility of combining measurements of a natural (as distinct from man-made) fallout radionuclide characterised by an essentially constant fallout input with those of ^{137}Cs whose input was effectively restricted to a period in the late 1950s and the 1960s, could offer additional scope for interpreting the erosional history of a study site. In addition, such a tracer could prove invaluable in areas where Chernobyl fallout has seriously complicated the interpretation of radiocaesium inventories. Equally, the availability of a fallout radionuclide with a much shorter half-life could offer potential for investigating the short-term behaviour and dynamics of erosion processes. Any other fallout radionuclide used in this manner in soil-erosion investigations would clearly need to be rapidly and strongly fixed on reaching the soil and two that would appear to offer such potential are ^{210}Pb and ^7Be . Figure 6, based on the work of Walling and Woodward [52], presents a typical example of the distribution of ^{137}Cs , ^{210}Pb and ^7Be in soils from both cultivated and pasture areas near Exeter, UK, which emphasises the very similar behaviour of the three radionuclides in terms of their fixation by the surface horizons of the soils. Some variations in their depth distributions are, however, evident and these reflect the different half-lives and the different fallout histories of the individual radionuclides. Further consideration of the potential for using ^{210}Pb and ^7Be in soil erosion investigations can usefully focus on each of these radionuclides in turn.

5.1. The use of unsupported ^{210}Pb in soil erosion investigations

Lead-210 is a natural product of the ^{238}U decay series, with a half-life of 22.26 years. It is derived from the decay of gaseous ^{222}Rn , the daughter of ^{226}Ra . Radium-226 occurs naturally in soils and rocks and will generate ^{210}Pb that will be in equilibrium with its parent. Diffusion of a small quantity of the ^{222}Rn from the soil introduces ^{210}Pb into the atmosphere and its subsequent fallout provides an input of this radionuclide to the soil surface that is not in equilibrium with its parent ^{226}Ra . This fallout component is commonly referred to as 'unsupported' or 'excess' ^{210}Pb , since it cannot be accounted for (or supported) by decay of the in-situ parent. The amount of unsupported or

atmospherically derived ^{210}Pb in a sediment sample can be calculated by measuring both ^{210}Pb and ^{226}Ra and subtracting the supported or in situ component. Recent advances in the use of low-background, low-energy gamma spectrometry [49, 50] make such measurements relatively easy to undertake.

Although ^{210}Pb has been extensively used for dating sediment cores [65, 66], its potential for use in soil erosion investigations has been essentially ignored to date. As a fallout radionuclide that is rapidly and strongly adsorbed by the surface soil, unsupported ^{210}Pb will behave in a manner similar to ^{137}Cs , except that its fallout input has been essentially constant through time and the supply to the soil surface is continuously replenished. This accounts for the slight difference in the depth distributions of the two radionuclides apparent in Fig. 6. Because of the continuous surface replenishment, unsupported ^{210}Pb concentrations are greatest at the surface, whereas in the case of ^{137}Cs , the lack of significant replenishment over the past 20 years, coupled with biological activity within the soil and some downward diffusion, have caused the level of maximum concentration to be displaced a few centimetres below the surface.

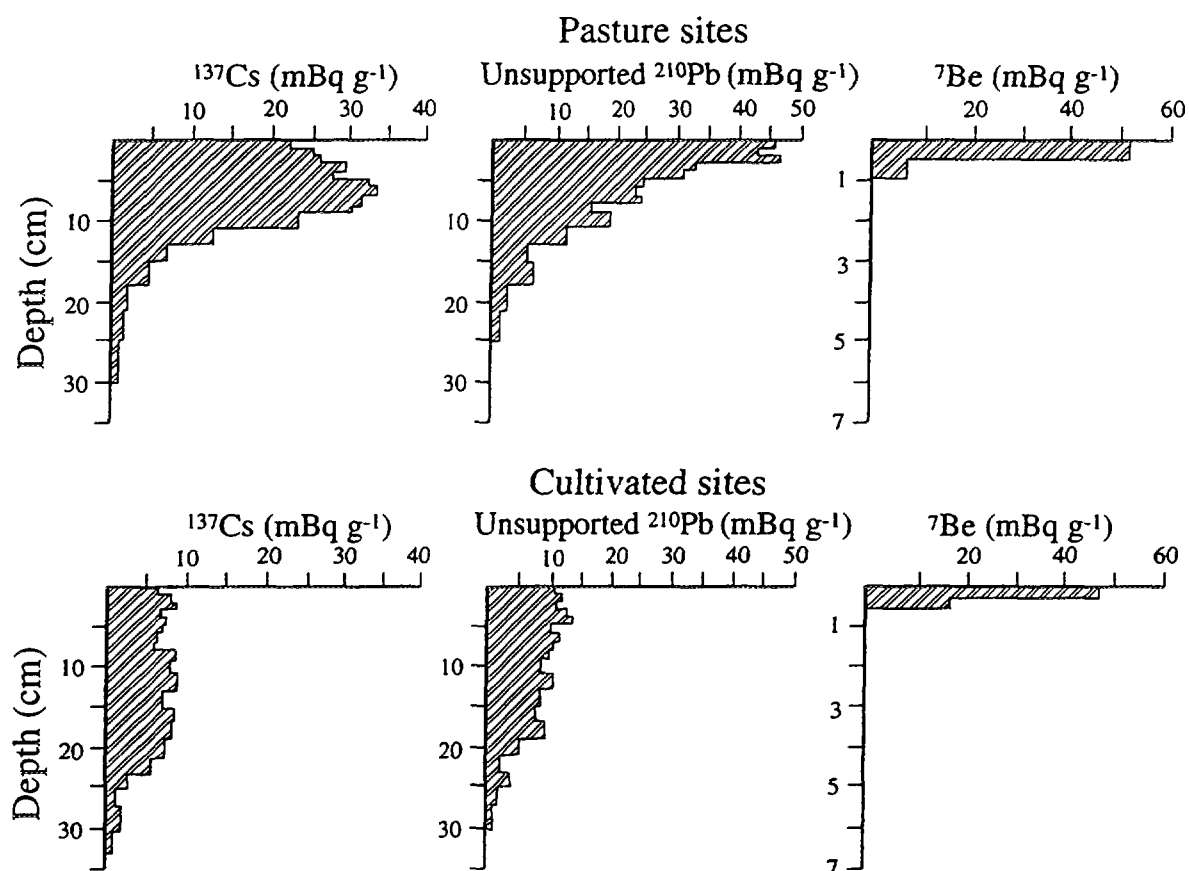


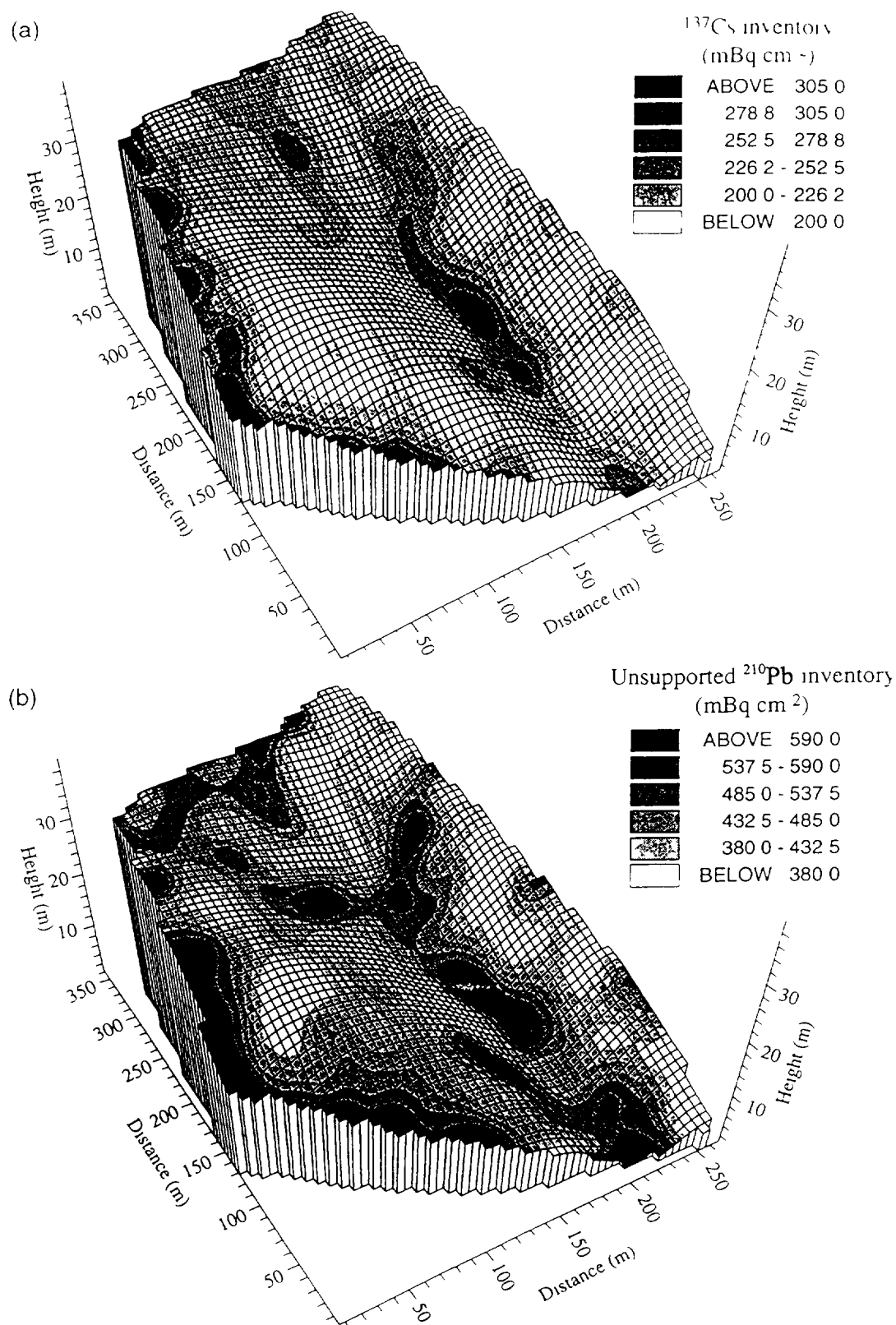
FIG. 6. Vertical distribution of ^{137}Cs , unsupported ^{210}Pb and ^7Be in soil profiles from representative pasture and cultivated sites near Exeter, Devon, UK.

Soil inventories associated with unsupported ^{210}Pb are generally somewhat greater than those of ^{137}Cs , and although little information exists as to the global pattern of unsupported ^{210}Pb fallout, there is some evidence that it is characterised by less variability than ^{137}Cs and that the existence of regions with very low inventories, where the approach may be difficult to apply, are likely to pose less of a problem.

Figure 7 emphasises the similar response of ^{137}Cs and unsupported ^{210}Pb to erosional redistribution within a small field at Buttsford Barton near Colebrooke, Devon, UK, investigated by Walling et al. [67, 68]. The maps presented are based on data assembled for a 20m x 20m grid of >200 soil cores. In this location the reference inventories for ^{137}Cs and unsupported ^{210}Pb were estimated to be ca. 275 mBq/cm^2 and 600 mBq/cm^2 respectively. Considering the field as a whole, the loss of ^{137}Cs and unsupported ^{210}Pb relative to their respective reference inventories amounts to 12.7 and 16.7% respectively. The increased loss and slightly greater range of inventory values associated with the unsupported ^{210}Pb measurements reflect the continuous input of this radionuclide and thus the annual potential for erosion of newly accumulated ^{210}Pb from the soil surface prior to incorporation into the soil by tillage and the longer period available for removal of this radionuclide, the input of which was not restricted to the period post the mid 1950s. The unsupported ^{210}Pb inventory values depicted in Fig.7b have been converted to estimates of the long-term soil redistribution rate by using a theoretically derived calibration relationship similar to that employed by Walling and Quine [15] to convert measured ^{137}Cs inventories to estimates of erosion and deposition rates. The result is presented in Fig. 8, which shows clear evidence of soil accumulation in the central depression that traverses the field and against the lower field boundaries. In addition, the influence of tillage in displacing soil from the slope convexities is also evident. Further scope undoubtedly exists for interpreting the relative magnitude of the ^{137}Cs and unsupported ^{210}Pb inventories for individual sampling points to provide information on the erosional history of the site. Whereas the latter will have been influenced by soil redistribution over the past 100 years or more, the former will reflect only soil redistribution that has taken place since the mid 1950s. Wallbrink and Murray working in Australia (personal communication) have shown that measurements of the ratio of unsupported ^{210}Pb to ^{137}Cs can also be used to estimate rates of soil loss, since the value of this ratio will vary down the soil profile in uncultivated soils and its magnitude can therefore be used to provide an indication of the proportion of the soil profile removed by erosion.

5.2. Use of ^7Be in soil-erosion investigations

Beryllium-7 is a cosmogenic radionuclide produced in the upper atmosphere by cosmic ray spallation of nitrogen and oxygen. In this case the radionuclide is extremely short-lived (half-life of 53.3 days) relative to ^{137}Cs and ^{210}Pb and it offers potential for investigating soil erosion dynamics over much shorter timescales. Figure 6 indicates that ^7Be is typically concentrated in the upper 5mm of the soil profile and is therefore capable of providing good discrimination between sediment derived from the immediate soil surface and that derived from depths > 10mm where concentrations will be near zero. Burch et al. [69] and Wallbrink and Murray [70,71] working in Australia were, for example, able to use this radionuclide to distinguish sediment mobilised by sheet erosion and rill erosion in erosion plot experiments, to investigate the progressive incision of rills during the course



of a single simulated runoff event and to provide an effective tracer of surface soil. Scope clearly also exists to couple measurements of ^7Be activity with equivalent measurements of ^{137}Cs and unsupported Pb^{210} to provide increased capacity for sediment source discrimination [52]. Monitoring of the spatial distribution of ^7Be activity across small plots immediately after storm events could also provide a basis for investigating local erosion patterns in response to microtopographical control.

6 PERSPECTIVE

Fallout radionuclides, particularly ^{137}Cs and Pb^{210} , have been extensively used for establishing the chronology of recent sediments, but their potential for use in soil erosion investigations has attracted much less attention. The growing need for information on the intensity of soil erosion in many areas of the world has, however, stimulated increasing interest in the use ^{137}Cs measurements to quantify rates and patterns of soil erosion and deposition. The validity and potential of this approach has now been demonstrated by numerous studies undertaken in a range of environments in many parts of the world, but there is still a need for further development, refinement and standardisation of the procedures used and for exploring additional applications.

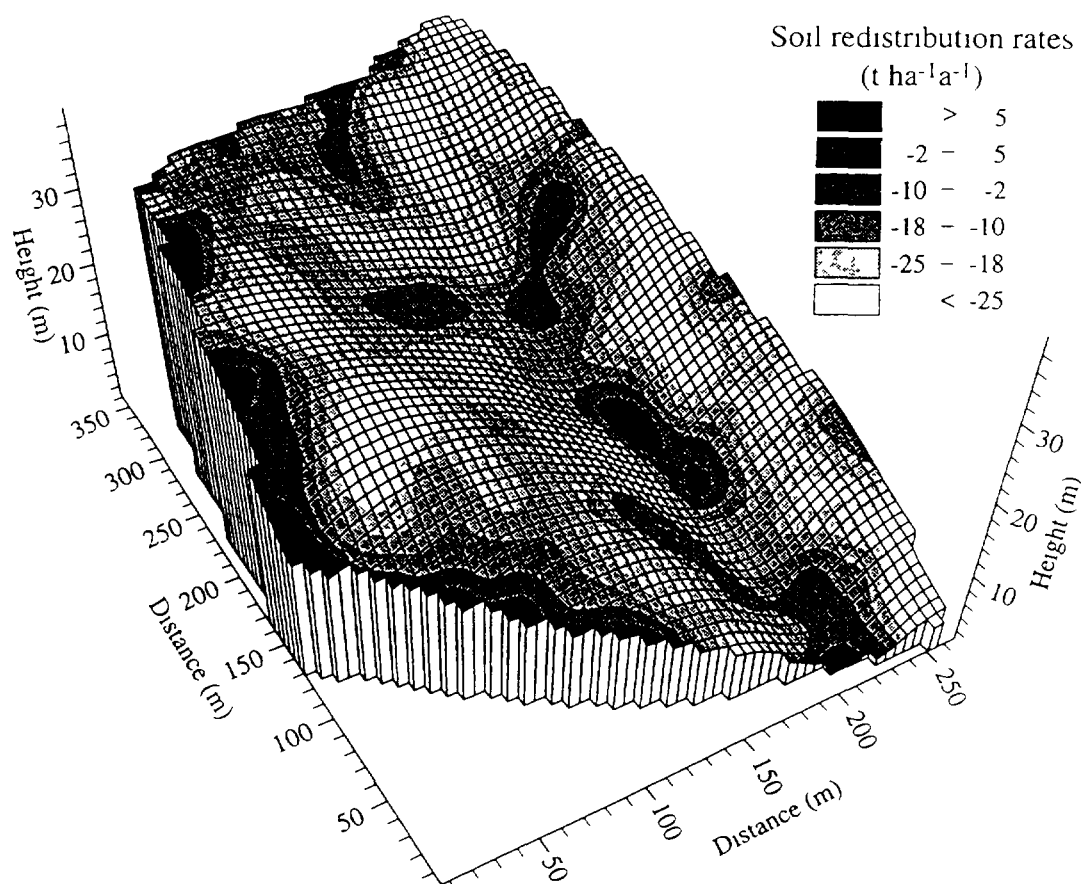


FIG. 8. Long-term soil redistribution rates within a field at Butsford Barton, near Colebrooke, Devon, UK, estimated from the unsupported ^{210}Pb inventory data presented in Fig. 7.

The use of other fallout radionuclides, including unsupported ^{210}Pb and ^7Be in soil-erosion investigations has attracted much less attention, but there is increasing evidence that they again offer considerable potential for use in soil-erosion investigations, both in their own right and as a complement to ^{137}Cs . Further work to explore and exploit this potential is clearly required

ACKNOWLEDGEMENTS

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BIBLIOGRAPHY¹ OF PUBLICATIONS OF ¹³⁷Cs STUDIES RELATED TO SOIL EROSION AND SEDIMENT DEPOSITION



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Abstract

BIBLIOGRAPHY OF PUBLICATIONS OF ¹³⁷Cs STUDIES RELATED TO SOIL EROSION AND SEDIMENT DEPOSITION.

This bibliography of some 1,600 citations presents significant publications as background for studies on soil erosion and sediment deposition. The ¹³⁷Cs technique, allows measurements of soil redistribution to be made quickly and efficiently, and we hope that this document will promote its use in planning strategies for landscape conservation.

1. INTRODUCTION

Soil erosion and its subsequent redeposition across the landscape is a major concern around the world. A quarter-century of research has shown that measurements of the spatial patterns of radioactive fallout caesium-137 can be used to measure soil erosion and sediment deposition on the landscape. By understanding the background for using the ¹³⁷Cs, scientists can obtain unique information to help in planning techniques to conserve the quality of the landscape. Research should continue on the development of the technique so that it can be used more extensively to understand the changing landscape.

On 16 July 1945 at 1230 Greenwich Civil Time, nuclear weapons tests were begun, releasing ¹³⁷Cs and other radioactive nuclides into the stratosphere. Over the 50 years since that first test, much research has been done to understand the movement and fate of ¹³⁷Cs in the global environment. Many of these studies are critical for understanding the application of ¹³⁷Cs to the study of soil erosion and the subsequent redeposition of the eroded particles on the land. This bibliography presents significant publications as background information for studies employing ¹³⁷Cs as a research tool, and includes citations on the application of ¹³⁷Cs technology to measure either soil erosion or sediment deposition.

¹Updated March 1, 1997

Although this collection is extensive, it is probable that some citations are missing. There has been a rapid increase in the number of publications on the use of ^{137}Cs in studies of erosion and sedimentation (Fig. 1), demonstrating the acceptance and widespread use the technique. We hope that this bibliography will further promote the technique, and assist those using or preparing to use ^{137}Cs in research on erosion and sediment deposition, for landscape preservation.

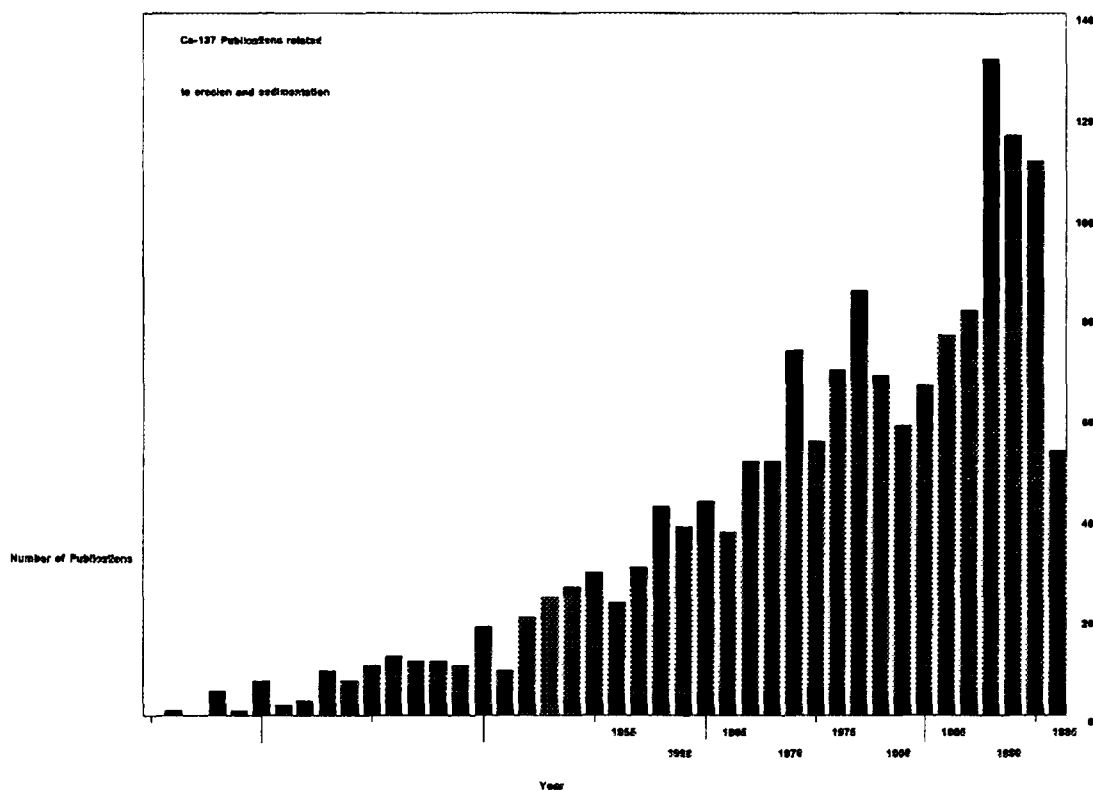


FIG. 1. Cumulative publications of ^{137}Cs related to soil erosion and sedimentation.

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GLOBAL DISTRIBUTION OF ^{137}Cs INPUTS FOR SOIL EROSION AND SEDIMENTATION STUDIES

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Abstract

GLOBAL DISTRIBUTION OF ^{137}Cs INPUTS FOR SOIL EROSION AND SEDIMENTATION STUDIES

A global distribution of ^{137}Cs deposition from the atmospheric nuclear tests, with estimates for 1996, excluding Chernobyl contribution, is presented, based on the global deposition data for ^{90}Sr . The data can be used to identify areas and countries, especially in the southern hemisphere, where the ^{137}Cs inventories are appropriate for soil erosion and sedimentation studies.

1. INTRODUCTION

Soil erosion is increasing all over the world, as a consequence of deforestation and inadequate agricultural practices. The negative impacts of soil erosion are twofold: loss of fertile top soil, which dramatically reduces food production yields, and environmental problems due to the transference of sediments to surface water bodies, which produces serious siltation problems in lakes and reservoirs.

The use of environmental ^{137}Cs for measuring soil erosion and deposition is well accepted. The average erosion or deposition rate over the last 3-4 decades can be estimated measuring the ^{137}Cs inventories in soil cores. Average siltation rate in lakes and reservoirs can also be estimated if the 1962 ^{137}Cs peak is noticeable in the vertical distribution of this radionuclide in bottom sediment core samples.

Unfortunately this technique can only be used in areas where the total ^{137}Cs deposition in the soil is great enough to allow its precise quantification, as the results are based on differences of concentration among samples. Very long counting times would increase the detection limit, but for practical reasons, counting times longer than 24 hours per sample are not convenient. The use of high efficiency detectors with Compton anticoincidence systems allows also the measurement of very low levels of ^{137}Cs , but their cost is too high for many developing countries.

To identify regions and countries of the world where the ^{137}Cs technique can yield good results for the evaluation of soil erosion and sedimentation processes, it is important to know the global deposition pattern of ^{137}Cs .

2. ACTIVITY OF ^{137}Cs INTRODUCED INTO THE ATMOSPHERE

^{137}Cs has been introduced into the atmosphere by the detonation of fission nuclear weapons. The activity produced is proportional to the weapon fission yield. It is assumed that the specific production rate for ^{137}Cs is 6.4×10^{15} Becquerel per megaton [1]. From the first atmospheric atomic explosion in Hiroshima in 1945 to the last in October 1980 in China, some 217.2 megatons of fission devices have been detonated in the atmosphere. The total activity of ^{137}Cs introduced into the atmosphere in that period can be estimated at 1400 PBq.

The deposition of this debris onto the surface of the planet is called fallout and it can be grouped into three categories: local, tropospheric and stratospheric. Local fallout, defined as the deposition within 100 miles of the test area, depends on the altitude and power of the detonation. If the fireball reaches the surface of the soil, higher amounts of local fallout are produced due to soil volatilisation. The local fallout consists of particles usually bigger than $50 \mu\text{m}$. [2, 3]. Due to the

remoteness of the test sites and to the relatively large particle size of the debris, this fraction is not suited for erosion studies. If local fallout is not considered, the total amount of ^{137}Cs globally dispersed from all the atmospheric nuclear tests is 912 PBq [4].

The distribution of the radioactive debris between troposphere and stratosphere depends on the power of the bomb. For nuclear bombs smaller than about 100 Kt, detonated in temperate latitudes, the radioactivity tends to remain in the troposphere. For detonations greater than 500 Kt, the debris is injected almost completely into the stratosphere [5].

In the stratosphere, at altitudes lower than 20 km, where most of the nuclear detonations have been carried out, the half-life for the transfer of aerosols between the hemispheres through the equator is about 60 months, while the half-life for transfer to the troposphere is only about 10 months, depending on the latitude [1]. Consequently, the bulk of the fallout from one test occurs over the hemisphere of injection.

The distribution of radioactive debris in the troposphere, either that introduced directly there or that transferred from the stratosphere, is governed by atmospheric circulation patterns. In each hemisphere there are two main air circulation cells, between the equatorial high pressure, sub-equatorial high pressure and sub-polar low pressure belts. At low latitudes, between the first two, the air near the surface circulates towards the equator, where it is heated and rises to an altitude of about 10 km. At this altitude, it starts to move horizontally towards the poles, coming down at latitudes of about 30 degrees, closing the first cell. At latitudes higher than 40 degrees, in the second cell, air near the surface moves northward, rising at about 60 degrees. At high altitudes, the air then moves in the direction of the equator, coming down to surface at a latitude of approximately 30 degrees. This circulation pattern explains why fallout is higher at medium latitudes, usually from 40 to 50 degrees and very low near the equator. The mean residence time of the dust introduced into the troposphere is on the average about 30 days [6].

3. ATMOSPHERIC REMOVAL PROCESSES

Fallout can be classified as dry fallout, which is removed from the atmosphere in dry weather, and wet fallout which is removed by precipitating weather conditions (rain or snow). Dry deposition is at least ten times lower than the wet fallout [7].

Rainfall removes aerosols from the troposphere mainly by droplet formation around the particle (rainout) and by scavenging (washout). This last process is not very effective, because the radioactivity concentration in rain remains almost constant along the rain episode [2]. For very small particles ($<0.01\mu\text{m}$), Brownian motion may play an important role for the transfer of aerosols to water particles, especially in clouds where small water droplets have a long residence time [2, 5]. The amount of radioactivity removed by the rain depends on the radioactivity concentration in the cloud. [2].

According to these considerations, the total amount of radioactivity deposited on the earth's surface depends on the atmospheric concentration of radioactivity and the amount of rainfall.

4. GLOBAL DEPOSITION OF ^{137}Cs

Radioactive fallout has been monitored all over the world since the early fifties. Studies performed in the late 40s identified ^{90}Sr as the most dangerous radionuclide in fallout, due to its biological fixation in bones and the consequent risk of developing bone cancer. Due to the significance of this radionuclide, its deposition has been monitored worldwide through two networks, one operated by the USA and another one by the UK. These data were used to elaborate a map with cumulative ^{90}Sr deposits in 1967 [8]. Unfortunately, ^{137}Cs deposition is not so widely documented.

It is possible, however to reconstruct the deposition pattern of ^{137}Cs using the ^{90}Sr data. The rate of production of these two nuclides (fission yields) is very well known. It has been proven that there is no fractionation between these radionuclides during the atmospheric transport [5]. The production rate for ^{137}Cs in fission weapon tests is 1.6 higher than for ^{90}Sr [4].

TABLE I. GLOBAL DEPOSITION OF Cs-137 (in Petabecquerels)

| Northern Hemisphere | | | | Southern Hemisphere | | | | Global |
|---------------------|--------|------------------------------------|-------------------------|---------------------|--------|------------------------------------|-------------------------|------------|
| Cs-137 deposition | | | | Cs-137 deposition | | | | cumulative |
| year | annual | cumulative (decay corrected) | yearly fraction % | year | annual | cumulative (decay corrected) | yearly fraction % | |
| 1954 | 4.7 | 5 | 0.59 | 1954 | 2.3 | 2 | 0.98 | 7 |
| 1955 | 20 | 25 | 2.53 | 1955 | 11 | 13 | 4.68 | 38 |
| 1956 | 31 | 55 | 3.92 | 1956 | 9 | 22 | 3.83 | 77 |
| 1957 | 30 | 84 | 3.80 | 1957 | 10 | 31 | 4.26 | 115 |
| 1958 | 40 | 122 | 5.06 | 1958 | 11 | 42 | 4.68 | 164 |
| 1959 | 80 | 199 | 10.13 | 1959 | 9 | 50 | 3.83 | 249 |
| 1960 | 18 | 213 | 2.28 | 1960 | 7 | 56 | 2.98 | 268 |
| 1961 | 21 | 229 | 2.66 | 1961 | 13 | 67 | 5.54 | 296 |
| 1962 | 92 | 315 | 11.65 | 1962 | 19 | 85 | 8.09 | 400 |
| 1963 | 150 | 458 | 18.99 | 1963 | 20 | 103 | 8.52 | 561 |
| 1964 | 100 | 548 | 12.66 | 1964 | 22 | 123 | 9.37 | 670 |
| 1965 | 34 | 569 | 4.30 | 1965 | 23 | 143 | 9.80 | 712 |
| 1966 | 19 | 575 | 2.41 | 1966 | 10 | 149 | 4.26 | 725 |
| 1967 | 9 | 571 | 1.14 | 1967 | 7 | 153 | 2.98 | 724 |
| 1968 | 12 | 570 | 1.52 | 1968 | 5 | 155 | 2.13 | 725 |
| 1969 | 7 | 564 | 0.89 | 1969 | 10 | 161 | 4.26 | 725 |
| 1970 | 7 | 558 | 0.89 | 1970 | 6 | 163 | 2.56 | 722 |
| 1971 | 9 | 554 | 1.14 | 1971 | 7 | 167 | 2.98 | 721 |
| 1972 | 4 | 546 | 0.51 | 1972 | 5 | 168 | 2.13 | 714 |
| 1973 | 3 | 536 | 0.38 | 1973 | 4 | 168 | 1.70 | 704 |
| 1974 | 5 | 529 | 0.63 | 1974 | 6 | 170 | 2.56 | 699 |
| 1975 | 4 | 521 | 0.51 | 1975 | 4 | 170 | 1.70 | 691 |
| 1976 | 2 | 511 | 0.25 | 1976 | 3 | 169 | 1.28 | 680 |
| 1977 | 4 | 503 | 0.51 | 1977 | 2 | 168 | 0.85 | 671 |
| 1978 | 4 | 496 | 0.51 | 1978 | 2 | 166 | 0.85 | 662 |
| 1979 | 2 | 487 | 0.25 | 1979 | 1.5 | 163 | 0.64 | 650 |
| 1980 | 1 | 476 | 0.13 | 1980 | 1 | 161 | 0.43 | 637 |
| 1981 | 3 | 469 | 0.38 | 1981 | 0.5 | 158 | 0.21 | 626 |
| 1982 | 1 | 459 | 0.13 | 1982 | 0.5 | 154 | 0.21 | 613 |
| 1983 | 1 | 449 | 0.13 | 1983 | 1 | 152 | 0.43 | 601 |
| 1984 | 0.5 | 440 | 0.06 | 1984 | 0.5 | 149 | 0.21 | 589 |
| 1985 | 0.5 | 430 | 0.06 | 1985 | 0.5 | 146 | 0.21 | 576 |
| 1986 | 70 | 490 | 8.86 | 1986 | 0.5 | 143 | 0.21 | 633 |
| 1987 | 0.5 | 480 | 0.06 | 1987 | 0.5 | 140 | 0.21 | 620 |
| 1988 | 0.5 | 469 | 0.06 | 1988 | 0.5 | 138 | 0.21 | 607 |
| 1989 | 0.5 | 459 | 0.06 | 1989 | 0.5 | 135 | 0.21 | 594 |

Adapted from Playford et al., 1990

The ^{90}Sr data obtained from the worldwide monitoring network of the UK have been transformed into deposition of ^{137}Cs for both hemispheres in Table 1. The yearly global deposition of ^{137}Cs and data for the decay-corrected cumulative deposition are included in this Table. The 70 PBq introduced in the northern hemisphere in 1986 from Chernobyl have also been included. Based on these data, it can be calculated that the amount of ^{137}Cs which was deposited on the earth surface until 1967 is 82% of the total amount in the northern hemisphere and 74% in the southern.

From the original map on the global deposition pattern of ^{90}Sr in mCi/km^2 , an equivalent one has been elaborated for ^{137}Cs , in mBq/cm^2 (Figure 1). The isolines correspond to the situation observed in 1967 but the values have been corrected taking into account the additional average deposition expected after that date, and the radioactive decay to 1996.

Some changes in the general patterns of the isolines could also be expected, due to the nuclear explosions by China and France, carried out after 1967 in the northern and southern hemispheres.

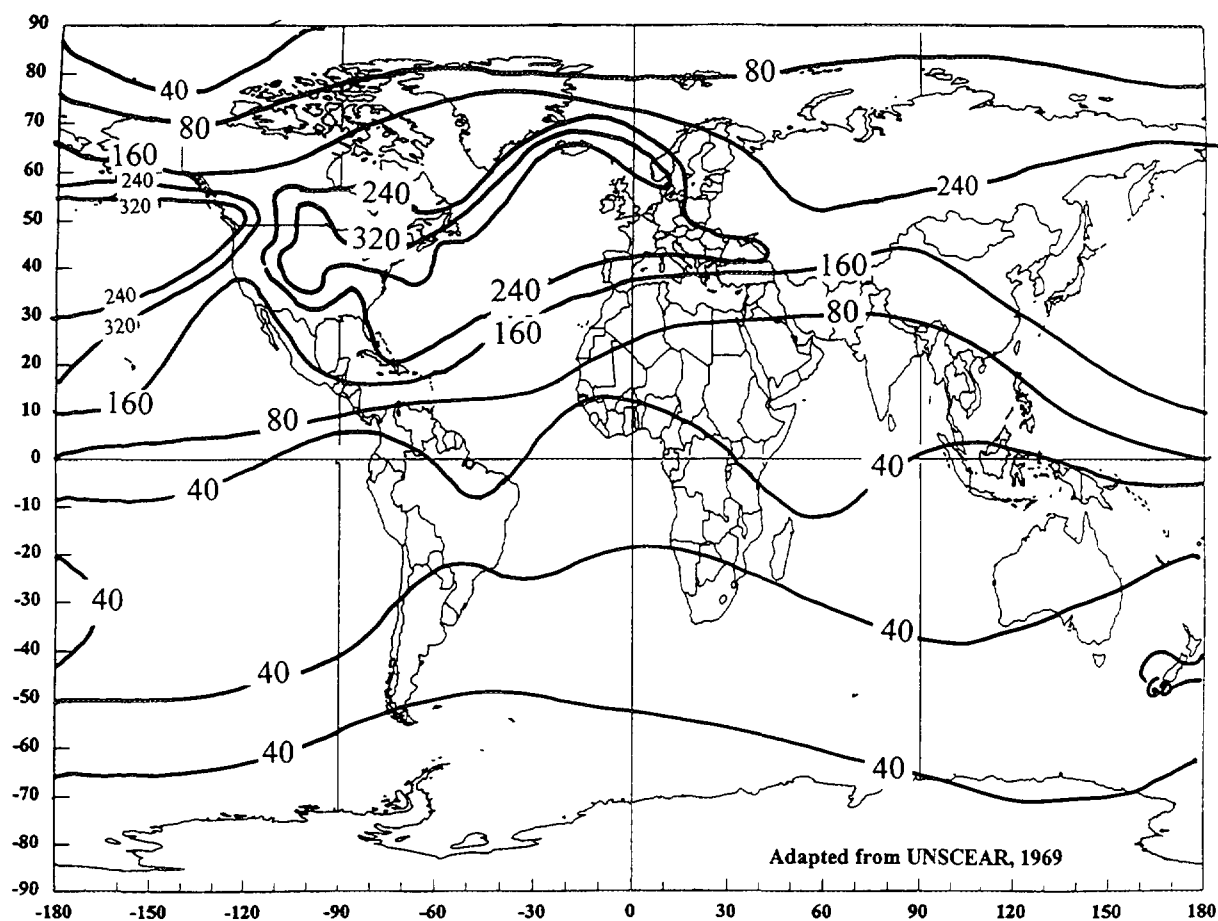


FIG. 1. Global inventory of ^{137}Cs from nuclear tests (in mBq/cm^2 , estimated for 1996). Adapted from UNSCEAR, 1969 [8].

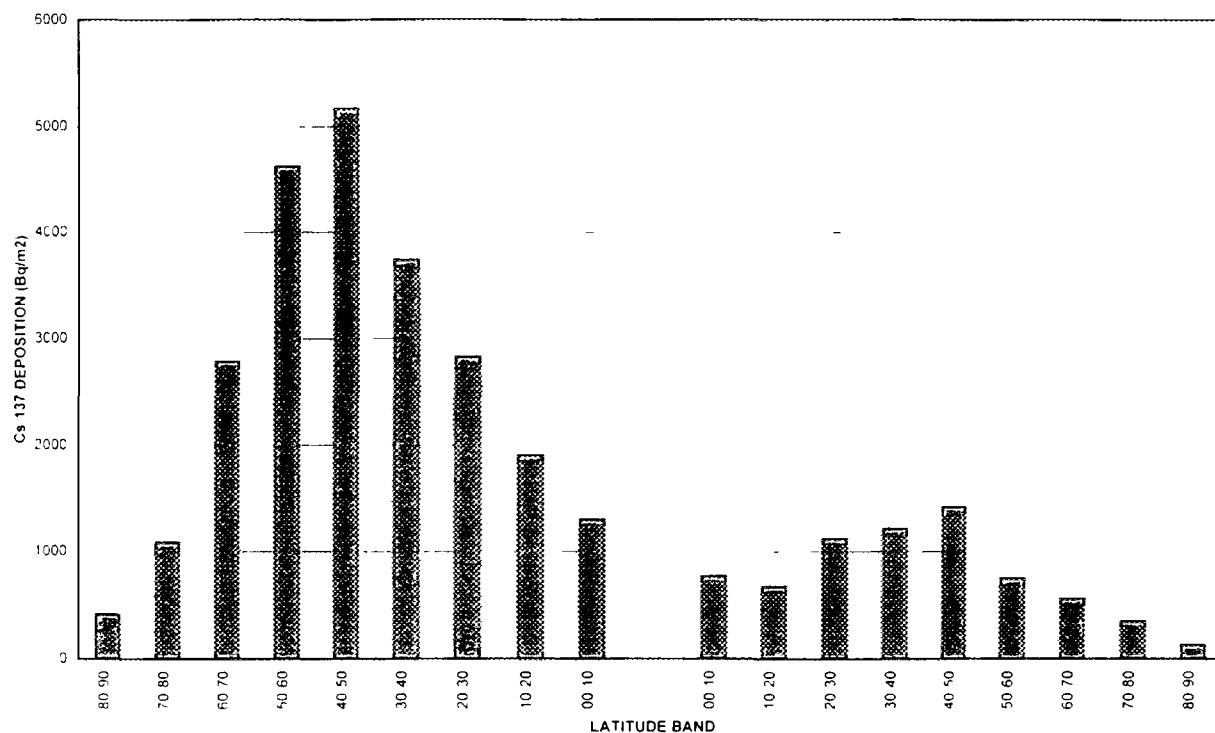


FIG. 2. Distribution of ^{137}Cs deposition by latitude bands.

However, as these two countries contributed with less than 10% to the total ^{137}Cs introduced into the atmosphere, these changes should not be very significant. On another hand, the Chernobyl accident produced important changes of the ^{137}Cs distribution pattern in some European countries. This fact is not reflected in Figure 1.

The distribution of ^{137}Cs deposition in the globe, by latitude bands, is presented in Figure 2. It has been elaborated with ^{90}Sr data presented in [4]. No further deposition has occurred since publication of the data. It can be seen that between the peaks in each hemisphere, the minimum deposition data are not in the equator but between 10° and 20° in the southern hemisphere.

5. CONCLUSIONS

The global ^{137}Cs deposition data were used to identify regions or countries, especially in the southern hemisphere, where the ^{137}Cs input would be appropriate for soil erosion studies. As such, regions in the southernmost parts of Africa (South Africa, Namibia and Botswana) and South America (south of Brazil, Uruguay, Argentina, Paraguay and Chile) seem to be adequate.

The potential of the ^{137}Cs technique for areas located near the equator has also been a matter of study. The northern part of South America, Central and West Africa, and Southeast Asia, should have enough ^{137}Cs in the soil for soil erosion studies. Countries in Africa and America located between latitudes 0 and 30°S should be excluded, as the ^{137}Cs inputs may be very low.

It is expected that additional data on ^{137}Cs inventories in soils will soon be provided and more reliable information on the global distribution of ^{137}Cs and its potential for the study of soil erosion and deposition in the southern hemisphere will be available.

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