QUANTIFICATION OF SOIL EROSION IN THE ALPS – MEASUREMENT AND MODELING

Inauguraldissertation

zur
Erlangung der Würde eines Doktors der Philosophie
vorgelegt der
Philosophisch-Naturwissenschaftlichen Fakultät
der Universität Basel

von

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Basel, 2010

Genehmigt von der Philosophisch-Naturwissenschaftlichen Fakultät auf Antrag von Dr. Ludwig Braun, Dr. Volker Prasuhn und

Prof. Dr. Ch. Alewell

Basel, den 26.05.2009

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CONTENT

ACKNOWLEDGEMENTS

Numerous people have contributed to the successful completion of this thesis in different ways. I would like to express my gratitude to each of them.

I would like to thank my lector Prof. Dr. Christine Alewell for supporting this interesting topic and for providing an open and friendly working environment.

I am very grateful to Dr. Volker Prasuhn who encouraged me and inspired me throughout all stages of my work. Sincere thanks goes to Claude Schneider, without him the technical realisation of the field measurement devices wouldn't have been possible. The local people in the Urseren valley supported me during my field work. I especially want to express my gratitude to Anton Simmen, Adrian Regli, Remo Christen and Bernhard Regli.

My colleagues in Basel, Dr. Katrin Meusburger and Dr. Monika Schaub made the three years an enjoyable time full inspiring discussions not only about scientific topics. Dr. Franz Conen and Dr. Dominik Baenninger always had an open door for my questions. A very special thank goes to Karin Liesenfeld for her administrative help.

My parents and my aunt Stephanie encouraged my interest in natural phenomena and enabled me to study such a fascinating subject. They further supported all my interests, whether they were related to my studies or to my spare-time activities.

Last but not least, I want to thank Markus, for sharing all my joys and frustration.

SUMMARY

Alpine regions have a high potential for soil erosion associated to extreme climatic and topographic conditions. Because of aggressive development in the recent past, environmental damage enhanced by acid deposition, global warming and development pressure of land use soil erosion in alpine areas has been an increasing concern to local, national and European policy makers (Francis, 1986; Schreurs, 2007; Yelpatyevskiy and Arzhanova, 1988). Numerous studies on soil erosion and erosion modeling were realized in lowlands or low mountain ranges. However, little is known about on- and off- site effects of soil erosion by water and snow melt in alpine terrain and the problem of quantification of these phenomena remains more or less unsolved. In mountain systems, difficulties in accessibility and data acquisition are paired with a high degree in small scale heterogeneity.

Thus, the main objective of this work was to evaluate appropriate soil erosion measurement procedures to use on alpine environments. Furthermore, the WEPP model (Water erosion prediction Project) and the USLE (Universal Soil Loss Equation) were run at the investigation sites. The aim was to assess both erosion prediction models on alpine environments since no suitable alpine model exists, so far.

Erosion measurements were done at three land use types with three replicates each. Land use types were hayfields (hf), pastures with dwarf shrubs (paw) and pastures without dwarf shrubs (pawo). These land use types represent the dominant land use types at the south facing slope in the Urseren Valley, were the sheet erosion takes place. The investigation sites are situated at an elevation of 1600-1800 meters a.s.l. The measurements to determine soil loss were done with sediment traps on plot measurements, sediment cups (point measurements) and with two Cs-137 based detection methods (point measurements). Cs-137 which is a common tracer for soil erosion in lowlands was adapted for application in high alpine environments. Cs-137 provides information about the spatial distribution and the extent of soil erosion in the investigation area. Soil erosion rates with Cesium-137 integrate the erosion since 1986, when Cesium-137 was released from the Chernobyl accident. A NaI in-situ spectrometer was calibrated for Cs-137 determination at steep mountain slopes. Calibration was done by comparing Cs-137 activities measured with GeLi detector in the laboratory and with NaI in-situ spectrometry at the same site. A close correlation between the two methods proved the validity of the in-situ measurements of the NaI detector system. Maximum monthly erosion rates during the vegetation periods 2007 and 2008 based on the sediment traps were 123 kg ha^{-1} for pasture without dwarf shrubs whereas minimum monthly erosion rates were obtained for pasture with dwarf shrubs and hayfields with 1 kg ha^{-1}. Sediment cups turned out to be a useful tool for point measurements. Additionally, cups can also be applied for soil erosion measurement during winter time. The measurements integrate over the whole wintertime, since slopes are not accessible. However, when the cups are filled, no quantitative statement can be done. Cs-137 based measurements based on in-situ detection lead to a maximum annual erosion rate since 1986 of 36 t ha^{-1} a^{-1} for hayfield hf1 and a minimum annual erosion rate of 8 t ha^{-1} a^{-1} for pasture with dwarf shrubs paw2. Pastures without dwarf shrubs have a mean annual erosion rate of 22 t ha^{-1} a^{-1} (s.d. 20%). Erosion values based on laboratory analyses with a GeLi detector were similar to erosion values from in-situ Cesium-137 measurements. R^2 of both measurement methods for all sites is 0.94. However, laboratory analyses need a soil sampling in the field. Since the alpine environment is very heterogeneous, especially on pasture sites, an extensive soil sampling is necessary to capture the full heterogeneity of erosion. But collecting big amounts of soil samples in the field does not seem adequate for sensitive mountain soils seriously affected by soil erosion.

Cesium-137 based erosion rates were compared with erosion rates predicted by the Universal Soil Loss Equation (USLE). The comparison was done in order to evaluate if the USLE is a

useful tool for erosion prediction in steep mountainous grassland systems. Erosion rates based on the USLE are in the same order of magnitude compared to Cs-137 based results for the land use type pasture with dwarf (predicted and measured erosion rates are between 4 and 12 t ha^{-1} a^{-1}). However, erosion amounts on hayfields and pasture without dwarf shrubs are underestimated by the USLE compared to Cs-137 based erosion rates. We assume that the underestimation is due to winter processes that cause soil erosion on sites without dwarf shrubs (e.g. snow gliding). The winter processes are not considered by the USLE. Dwarf shrubs may possibly prevent damage of soil erosion through winter processes.

In addition to the USLE we tested the WEPP model (Water Erosion Prediction Project) to describe the soil erosion in the Urseren Valley as it seems to be one of the most promising models for steep mountainous environments. Crucial model parameters were determined in the field (slope, plant species, fractional vegetation cover, initial saturation level), by laboratory analyses (grain size, organic matter) or taken from the WEPP manual (soil erodibility, effective hydraulic conductivity, cation exchange capacity). Erosion rates were measured with sediment traps during the vegetation period between June 2006 and November 2007. Long-term soil erosion rates were estimated by measuring Cs-137 redistribution as described above. In addition to the erosion rates, soil moisture and surface flow was measured during the vegetation period in the field and compared to model outputs. Short-term erosion rate simulations for the vegetation period in 2007 are in agreement with measured erosion rates (predicted and measured erosion rates are between 0 and 0.4 kg ha^{-1} mo^{-1} for hf3, between 0 and 3.4 kg ha^{-1} mo^{-1} for pawo2 and between 0 and 1.1 kg ha^{-1} mo^{-1} for paw2). However, simulated soil moisture is up to two times higher than measured field data. Furthermore, simulated soil moisture is increasing during spring time while measured soil moisture is decreasing during the same time and surface flow is not simulated correctly. Snow cover melting is simulated too late compared to field observations and thus water from snowmelt is available until summer time in 2007. We assume that these differences lead to the general underestimation of erosion rates for long-term rate erosion predictions for all three land use types. Thus, the WEPP model could be a useful tool for alpine regions during the vegetation period to assess the influence of different land use conditions but should be applied carefully during winter time and on snow covered regions. Generally, neither WEPP nor USLE contain avalanches and snow gliding processes. The Cs-137 based measurement rates point out that winter processes seems to be important for high erosion rates during longer time periods.

Our study demonstrates the need of soil conservation strategies in alpine regions since erosion rates are much higher than previously reported. Furthermore, results of the WEPP model are only comparable during the vegetation period with measured data on respective slopes. Also, the accuracy of USLE results is not satisfactory on the affected sites. Thus, a first attempt was done to create an alpine factor for the USLE based on the measured data. Hence, existing models have to be adapted to alpine regions or new soil erosion models have to be designed for steep mountainous slopes.

CHAPTER 1

Introduction

The European Alps are the most intensively exploited mountain region in the world, inhabited by 13.6 Mio people and visited by ca. 120 Mio visitors every year (Bätzing, 1997). Primary resources are agricultural production, landscape values (e.g. tourism, transportation corridors) and hydroelectric power. Human pressure on the alpine environment has increased since the beginning of the 1970s (Isselin-Nondedeu and Bedecarrats, 2007). The extreme topography and climate result in high instability, fragility and sensitivity of these ecosystems. While soil erosion is a well studied phenomenon in lowlands (especially in arable land), only a few studies about soil erosion in alpine grasslands have been done so far. Furthermore, soil erosion quantification under natural precipitation regimes are scarce (Descroix and Mathys, 2003; Felix and Johannes, 1995). Thus, soil erosion in the Alps was identified as a priority for action by the soil protocol of the Alpine Convention (AlpineConvention, 2005), but a comprehensive assessment of soil erosion of the Alps is still missing (ClimChAlps, 2006). Erosion processes in alpine regions differ from lowland erosion in many aspects: the soils are less developed, they are more intensively exposed to freezing-thawing and snow-cover processes, they are exposed to extreme climate and topography and often a high infiltration rate which results in moderate overland flow. Simultaneously, observed damages are very high but experimental and theoretical methods suitable for alpine terrain are not available. A critical evaluation of well-established methods to determine soil erosion in mostly arable crop soils is crucial for steep alpine environments, because reliable information on soil erosion rates is an essential prerequisite for the design of targeted erosion and sediment control strategies.

Moreover, soil erosion assessments for the Alps are based on models developed for lowlands and often lack of serious validation. This validation is necessary to evaluate strengths and lack of strengths of existing models.

Several projects concerning soil erosion in alpine areas have been done at the University of Basel. One project addressed the qualitative approaches with stable isotopes such as carbon, nitrogen and oxygen whereas another project treats the problem of soil erosion risk assessment in the Alps on catchment scale with remote sensing and GIS tools.

The aim of this project was to identify suitable methods to measure soil erosion on steep alpine sites. Furthermore, it was clarified if there are soil erosion prediction models for arable land that are also suitable to use in alpine regions. For the measurement of soil erosion, sediment traps, sediment cups and Cs-137 were used (Chapter 2). The Cs-137 in-situ method was adjusted to alpine regions since the laboratory analyses that are most common, is a destructive method especially in alpine regions (Chapter 3 and 5). Two different models were applied in the Urseren Valley: the empirically based model USLE (Universal soil loss Equation, Chapter 3) and the WEPP model (Water Erosion Prediction Project, Chapter 4). Chapter six addresses the application of the USLE in the Urseren Valley with improved information on fractional vegetation cover. The data were compared to Cs-137 based erosion values.

1.1 Field installations for soil erosion measurement

Mapping and quantification of soil erosion under different land use conditions has been studied in numerous projects for agricultural soils in lowlands or low mountain ranges (e.g. Gabriels et al., 2003; Leser et al., 2002; Nearing et al., 1999; Prasuhn et al., 2007). Since the influence of snow makes it difficult to measure soil erosion in alpine regions around the year, most of the erosion measurements in alpine regions have been done during the vegetation period without the influence of winter processes (Descroix and Mathys, 2003; Felix and Johannes, 1995; Isselin-Nondedeu and Bedecarrats, 2007). Erosion measurements over several years that also included winter processes were done with USLE test plots by Frankenberg et al. (1995). So far, the classical methods to measure soil erosion were done either with sediment traps (e.g. Pieri et al., 2007; Robichaud and Brown, 2002), with erosion pins (e.g. Haigh, 1977; Hancock et al., 2008), splash cups (e.g. Mati, 1994; Van Dijk et al., 2003), Coshocton wheels of plot and ha measurement (e.g. Bonta, 2002; Rochester et al., 1994), USLE test plot measurements (e.g. Bagarello et al., 2008; Wischmeier and Smith, 1978a) and with radioactive isotope measurements such as Caesium-137, Beryllium-7 and Lead-210 (e.g. Matisoff et al., 2002b; Walling et al., 1999).

1.2 Cesium-137 based soil erosion quantification

Cs-137, Pb-210 and Be-7 have been used before as tracers for soil erosion. Be-7 (46.5 keV) with its half life time of 53.12 days is only suitable for measurement of recent erosion processes. Pb-210 (477.6 keV) has a half life time of 22.3 years. The determination of Be-7 and Pb-210 needs a good resolution of the specific energy lines (given in keV). Because of the good peak resolution in spectra measured by Ge detectors, these are usually favored over the NaI detectors. However, the use of a Ge detector in the field is difficult as Ge detector systems are usually relatively heavy or not portable at all because of the Ge-crystal's need for cooling. Thus, a NaI detector system was calibrated for Cs-137 measurements at steep mountain slopes (chapter 5). Among the three isotopes Be-7, Pb-210 and Cs-137, Cs-137 is the most commonly used in erosion studies because it is relatively easy to measure, has a well defined date of input and with its half life of 30.17 years provides information about medium-term erosion (Ritchie and McHenry, 1990). Cs-137 is an artificial nuclide and has its origin either in bomb-testings in the 1960ies and 70ies or in the Chernobyl reactor accident in 1986. For the Urseren Valley, about 90% of Cs-137 was deposited after the Chernobyl reactor accident (source: Federal Office of Public Health, unpublished). Therefore, the detection of Cs-137 concentration in the Urseren Valley provides important information on soil erosion since 1986. After deposition Cs-137 is rapidly and tightly bound to the fine particles in the soil, i.e. clay minerals and organic matter. Movement by chemical and biological processes are strongly limited (Ritchie and McHenry, 1990). Redistribution is mainly caused by physical processes where Cs-137 moves with soil particles (e.g. Bonnett, 1990b; Ritchie and McHenry, 1990). The vertical distribution of Cs-137 in cultivated soils is influenced by the tillage practice which results in a more or less homogenous Cs-137 activity within the plough layer (He and Walling, 2000; Owens et al., 1996; Ritchie and McCarty, 2003b). In unploughed soils most of the Cs-137 is accumulated at the top of the soil profile or few centimeters below and the content decreases exponentially with depth (Owens et al., 1996; Ritchie and McCarty, 2003b; Ritchie and McHenry, 1990). Soil erosion and redistribution processes can therefore be tracked down by the measurement of the Cs-137 activity. Various studies applied the Cs-137 method to track down soil erosion. However, mainly for agricultural areas in different parts of the world (for overview see Wicherek and Bernard, 1995) and mostly with Ge detector measurements in the laboratory (e.g. Walling et al., 1999).

It is possible to distinguish areas of net soil loss from net deposition areas by analyzing the spatial distribution of Cs-137 in the studied area (Ritchie and McHenry, 1990). However, applicability of the method depends on the Cs-137 activity in the catchment as not all parts of Switzerland were impacted with measurable amounts of Cs-137 after the Chernobyl reactor accident (Hofmann et al., 1995). Cs-137 activity in the Urseren Valley is high enough for soil erosion measurements because defined energy peaks are detectable (Schaub et al., 2009).

1.3 Soil erosion prediction based on USLE and WEPP

The Universal soil loss equation USLE (Wischmeier and Smith, 1978a) is an empirically based model to identify erosion risk areas and to help in decision making processes for long term average erosion development in agricultural lowlands (e.g. Auerswald et al., 2003). Erosion rates are calculated as

$$A = R\ K\ L\ S\ P\ C \tag{1.1}$$

where A is the average annual soil loss (kg m^{-2} a^{-1}) due to erosion, R is the rainfall-runoff factor (N h^{-1}), K is the soil erodibility factor and gives the soil's tendency to erode (N^{-1} h kg m^{-2}). The topography factor combines slope length S (-) and slope steepness L (-), P is the support practice factor (-) and C is the cover and management factor (-).The USLE was originally parameterized for soil erosion evaluation in the United States. Modified versions for other regions as well as for different temporal resolutions have been developed in the past. These models are for example the German USLE that is called ABAG and the revised version of the USLE, the RUSLE (Revised Universal Soil Loss Equation with a higher temporal resolution than USLE). For our study, single factors of these three models were chosen depending on the best suitability for our site. The factors were either determined from field data or taken from literature values. Thus, we did not take the factors as proposed by Wischmeier and Smith (1978) for the USLE but tried to get the most appropriate factor since no special model is designed for mountainous regions.

The Water Erosion Prediction Project (WEPP) is a frequently used tool to simulate water erosion and sediment yield. WEPP has been tested and applied in various geographic locations across the United States (Huang et al., 1996; Laflen et al., 2004; Savabi, 1993), in Australia (Yu and Rosewell, 2001) and in Europe (Brazier et al., 2000; Gronsten and Lundekvam, 2006; Pieri et al., 2007; Raclot and Albergel, 2006). The application of WEPP in steep alpine environments, has been tested only once in the Italian Alps by Simonato et al. (2002). This study resulted in a relatively good simulation of erosion rates compared to collected soil erosion data (Simonato et al., 2002). However, hydrological parameters were not measured. Hence, the overall quality of the hydrology and of the total model output could not be verified. For our study the WEPP model was chosen because it describes separately and in detail plot size, cattle trails, vegetation and fractional vegetation cover, precipitation amount and intensities, land use type and snow processes (snow accumulation and snow ablation). Thus, it covers many processes that are essential for alpine regions.

1.4 Aims of the project

An Alpine Convention was established in 1989 stipulating that the contracting parties (Germany, France, Italy, Slovenia, Lichtenstein, Austria, Switzerland and the European Community) should pursue a comprehensive policy for the preservation and protection of the Alps (Montanarella and Nègere, 2001). However, recent research on soil erosion and runoff has mostly focused on lowland environments with agriculturally productive soils, where erosion is a threat to economic values. For the promotion of agro-environmental indicators, conservation practices and sustainable use of the soil resources the geomorphology as well as the specific environment of the mountains has to be taken into account. Information of alpine soils is urgently needed to be able to protect these environments in a sensible way. Within this project we aim to move forward in closing information gaps on the determination of soil erosion rates as well as the modeling of soil erosion in the Swiss Alps quantitatively. The aim of our work was to evaluate the most appropriate method for soil erosion measurement in steep alpine environments. Thus, the decision should be simplified which measurement method can be used for soil erosion quantification in the future. For this reason, different types of soil erosion measurements that were used in the past were applied and evaluated. Furthermore, the intension within this work was to identify the influence of snow processes on soil erosion in comparison to erosion processes during the snow free period. Additionally, erosion rates of different land use types were considered. Pastures were subdivided into pastures with and without dwarf shrubs. Additionally, since no erosion prediction model for steep alpine environments is available, we tested the USLE as it is one of the most used erosion models worldwide. Cs-137 based erosion rates were compared to modeled results of the USLE (A) in order to assess the suitability of the model for steep alpine grasslands. The physically based model WEPP (Water Erosion Prediction Project) was the second model that was applied for the Urseren Valley. Output of overland flow, soil moisture, erosion rates as well as snow height were compared to measured data in the field.

CHAPTER 2

ON THE MEASUREMENT OF ALPINE SOIL EROSION ON PLOT SCALE

Submitted to: Catena Special Issue: Experimental Methods in Soil Erosion Studies
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2.1 Summary

The knowledge of soil erosion processes and especially soil erosion rates in alpine grassland regions is scarce because only a few studies on soil erosion measurements have been done, so far. We distinguished between sediment traps and sediment cups to determine erosion rates during the vegetation period and Cs-137 based measurements to measure long term erosion rates since 1986. The latter method integrates over a time span of 22 years, thus including erosion rates during winter time. We investigated three different land use types: hayfields, pasture with dwarf shrubs and pasture without dwarf shrubs in the Urseren Valley (Central Switzerland) with a mean slope steepness of 37°. Mean monthly erosion rates during the vegetation periods 2007 and 2008 based on the sediment traps were 45 kg ha^{-1} mo^{-1} (s.d. 56 t ha^{-1} a^{-1}) for pastures without dwarf shrubs, 12 kg ha^{-1} mo^{-1} (s.d. 12 t ha^{-1} a^{-1}) for hayfields and 6 kg ha^{-1} mo^{-1} (s.d. 7 t ha^{-1} a^{-1}) for pastures with dwarf. These generally low erosion rates can be explained by a low overland flow during the vegetation period of 0.5-1.8% of the measured precipitation. Cs-137 based measurements yielded mean annual erosion rates for the time span 1986 - 2008 of 26 t ha^{-1} a^{-1} (s.d. 14 t ha^{-1} a^{-1}) for hayfields and annual erosion rates of 8.3 t ha-1 a-1 (s.d. 2.5 t ha^{-1} a^{-1}) for pastures with dwarf shrubs. Pasture without dwarf shrubs have mean annual erosion rates of 24 t ha^{-1} a^{-1} (s.d. 4 t ha^{-1} a^{-1}). Cs-137 based erosion rates exceeded sediment traps by a factor 200. We conclude that erosion rates during the vegetation period are only about 2% of the total mean annual amount of erosion in the investigated alpine grassland systems.

2.2 Introduction

Erosion is a formative geomorphologic process in alpine environments due to steep slopes and extreme climate. Rock falls, avalanches and landslides are among those formative processes and have been studied many times to prevent from human damage (e.g. Fell et al., 2008; Oppikofer et al., 2008; e.g. Wang and Cavers, 2008). However, the investigation of soil erosion on alpine sites was limited to a few studies, so far. The term soil erosion is used for sheet, rill, interrill and gully erosion as well as for landslides. Alpine grasslands that are the focus within this study do not have the typical rill and interrill pattern. Rough surfaces occur on the grassland but continuous rills down slope do not exist. This is the major difference to soil erosion on arable land. Here, we focus on sheet erosion that is defined as erosion caused by surface water in unconcentrated flow. Soil erosion and the regeneration of soil has been mainly studied on ski slopes (e.g. Leser and Mosimann, 1982). Soil erosion in the Alps is a well recognized problem, identified as a priority for action within the soil protocol of the Alpine Convention (e.g. AlpineConvention, 2005). However, a comprehensive assessment of soil erosion in the Alps is still missing (ClimChAlps, 2006) and the knowledge of soil erosion and especially sheet erosion on alpine grasslands remains scarce. For agricultural sites in lowlands of low mountain ranges, mapping and quantification of soil erosion under different land use conditions has been studied comprehensively (e.g. Gabriels et al., 2003; Ledermann et al., 2008; Leser et al., 2002; Matisoff et al., 2002a; Nearing et al., 1999). Generally, different erosion processes operate at different temporal and spatial scales and measurements must be adapted to the scale (Stroosnijder, 2005). Stroosnijder (2005) defined five relevant spatial scales for water erosion in agricultural systems: (1) the point scale (1m2) for interrill (splash) erosion, (2) the plot (<100 m2) for rill erosion, (3) the hill slope (< 500 m) for sediment deposition, (4) the field (<1 ha) for channels and (5) the small watershed (<50 ha) for spatial interaction effects. The measurement methods of soil erosion on point scale are splash cups (e.g. Mati, 1994; Van Dijk et al., 2003). On plot scale sediment traps (e.g. Pieri et al., 2007; Robichaud and Brown, 2002), Coshocton wheels (e.g. Bonta, 2002; Rochester et al., 1994) and USLE test plot are used (e.g. Bagarello et al., 2008; Wischmeier and Smith, 1978). Radioactive isotope measurements such as Caesium-137, Beryllium-7 and Lead-210 (e.g. Matisoff et al., 2002b; Walling et al., 1999, Mabit, et al. 2007) have been used for point measurements that can be extrapolated to plots, depending on the heterogeneity and the number of measurements on the site. The Cs-137 based erosion measurement is a common method on arable land that has been used many times (e.g. Walling, (2004)). In alpine regions, however, the Cs-137 method was not used before to quantify soil erosion with the exception of measurements within this project in the Urseren Valley (Konz et al., 2009b). Furthermore, in-situ as well as laboratory Cs-137 measurements based on soil sampling were done to test the feasibility of those methods in alpine regions (Schaub et al., 2009). A detailed discussion of general advantages and disadvantages on field and laboratory applications of gamma detectors can be found in Beck et al. (1972), Miller & Sebell, (1993) and He & Walling, (2000).

Regarding measurements during the vegetation period, some of the plot measurement methods have been also sporadically used in alpine environments (e.g. Felix and Johannes, 1995). But since snow dynamics make it impossible to measure soil erosion in alpine regions throughout the year, erosion measurements in alpine regions have been done during the vegetation period only without the influence of snow (Descroix and Mathys, 2003; Felix and Johannes, 1995; Isselin-Nondedeu and Bedecarrats, 2007). Felix and Johannes (1995) measured erosion rates between 0.1 and 200 kg ha^{-1} during the vegetation period with sediment traps. Isselin-Nondedeu and Bedecarrats (2007) determined the influence of several plants on soil erosion. They found considerable differences between plant species with Festuca Alpina having the highest amount of sediment deposition. Frankenberger et al. (1995) measured erosion rates up to 20 t $ha^{-1}a^{-1}$ on arable alpine sites during the vegetation period.

The aim of our work was to measure soil erosion rates on nine plots in an alpine environment. Since slope steepness, land use conditions as well as vegetation cover have to be constant, the measurement plots were planned to be 2 meters in with and 20 meters in length. The results should provide further information on soil erosion rates of alpine sites since only scarce information exists. A further aim of our study was to evaluate the three applied measurement methods in respect to their suitability for alpine regions. The measurements were done at three different land use types (hayfields and pastures, whereas pastures were subdivided into pastures with and without dwarf shrubs) with three replicates each to determine the main land use types in the Urseren Valley. The valley was chosen because of the great damages due to soil erosion at the south facing hillslopes. Measurements were conceived to give separate information on erosion rates during the vegetation period (sediment traps) and whole year rates (Cs-137 measurements) to separate out influence of winter processes. The information of soil erosion results from sediment traps were supplemented by point measurements with sediment cups to collect point information in addition to the plot measurements.

2.3 Materials and Methods

2.3.1 Investigation area

The study area is located in Central Switzerland (Canton Uri) in the Urseren valley (Figure 2.1). The elevation of the W-E extended mountain valley ranges from about 1400 m a.s.l. up to about 2500 m a.s.l.

The mean annual rainfall from 1986 to 2008 is 1516 mm, mean air temperature is 3.1 °C (MeteoSwiss, 2007). The valley is snow covered from about November to April with the maximum snow height in March (Ambuehl, 1961) and a mean annual snowfall from 1986 to 2008 of 448 mm. Surface flow is usually dominated by snowmelt from May to June. Important contributions to the flow regime are early autumn floods. The dominant land use types in the valley are hayfields near the valley bottom (from 1400 to approximately 1850 m.a.s.l.) and pasture further upslope. Siliceous material is dominant, forming at our sites cambic podzols (anthric) and podzols (anthric) classified after IUSS Working group (2006). The characteristic of these soils is a migration (M-horizon) horizon within the upper 100 cm that has been caused by sedimentation in the past. The thickness of the M-horizon is between 5 and 45 cm. For a detailed description of the Urseren Valley see (Meusburger and Alewell, 2008).

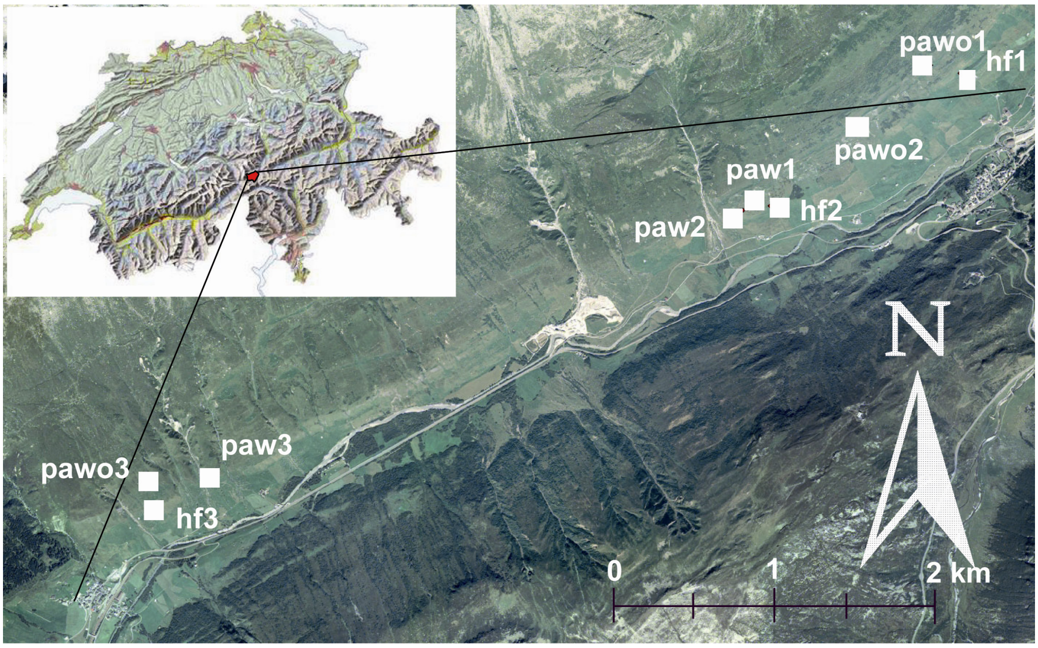

Figure 2 1: The Urseren Valley in Southern Switzerland and the location of the investigated sites with three grassland types: hayfield (hf), pasture without dwarf shrubs (pawo) and pasture with dwarf shrubs (paw).

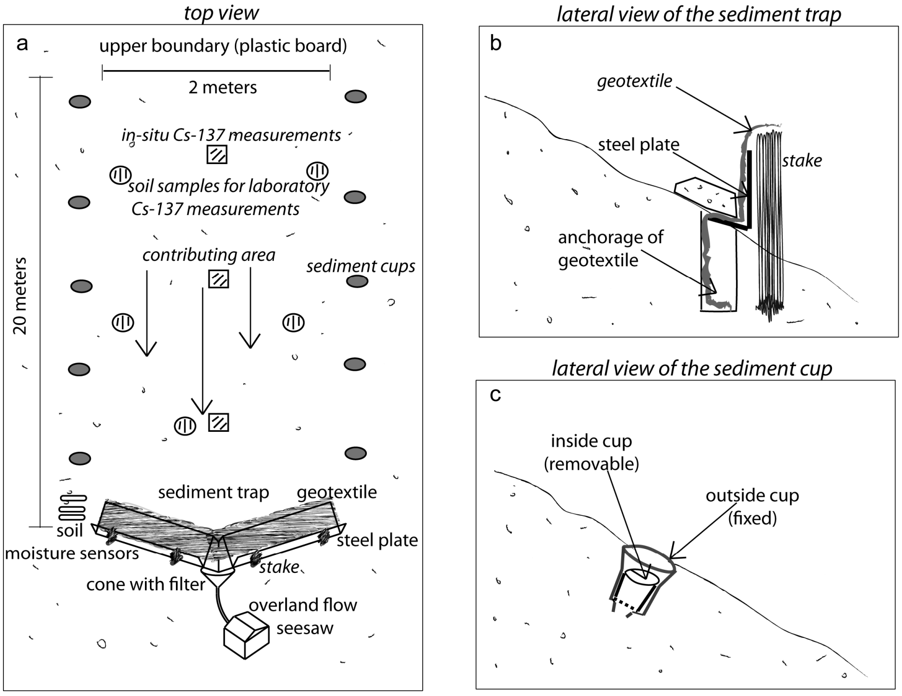

Figure 2.2: Composition of all soil erosion measurements with sediment traps, sediment cups and Cesium-137 measurements as well as overland flow and soil moisture measurements (a) in the Urseren Valley. A lateral view of sediment traps (b) and sediment cups (c).

2.3.2 Experimental plots

The nine experimental plots are situated at the south-facing slope at an altitude of 1550 m a.s.l up to 1800 m a.s.l. Three different land use types with three replicates each were investigated: hayfields, (hf 1-3), pasture with dwarf shrubs (paw 1-3) and pasture without dwarf shrubs (pawo 1-3). The slopes of all plots were in the range of 35°-39°. Soil type of hayfield hf2, paw2 and pawo1 is sandy loamy silt, paw1 is loamy sand and hf1, hf3, pawo2, pawo3 and paw3 is silty loamy sand. Vegetation of hayfields is dominated by Trifolium pratense ssp. Partense, Festuca sp., Thymus serpyllum and Agrostis capillaries. Pasture with dwarf shrubs are dominated by Calluna vullgaris, Vaccinium myrtillus, Festuca violacea, Agrostis capillaries and Thymus serpyllum. Dominant vegetation type at pasture without dwarf shrubs are Glubelaria cordifolia, Festuca sp. and Thymus serpyllum. Experimental plots of all pastures are dominated by horizontal cattle trails. The investigation sites are not within or right beside a landslide but clearly separated to avoid mixture of information especially from Cs-137 measurements. All slopes are separated from upper slopes by large terraces. Pastures are stocked by cattle from June till September. Stocking rates on single sites are not fixed and may change within different years. A flock of sheep cross the valley at the beginning and the end of the vegetation period for all land use types.

Sediment traps

Sediment traps were installed on each plot in July 2006 using a geotextile which is fixed to the ground (Figure 2.2b). The construction was carried out based on Robichaud and Brown (2002). The sediment trap after Robichaud and Brown (2002) was extended by means of a v-shaped steel plane below the geotextile to concentrate and to measure the surface water flow (Figure 2.2b). Water and soil that flushed into the geotextile was collected every second week during the vegetation period from April to November. The contribution area of the sediment trap is 40 m2 (2 meters in width and 20 meters in length). The upper boundary was defined

trough plastic boards that were plunged 20 cm into the ground and 20 cm overlapped at the upper side. The plots were chosen in a way that the topography clearly defines the side boundaries. That means that neither concave nor convex slopes were chosen. Furthermore, side boundaries were not considered since it cause additional disturbance in stony soils.
In addition to the sediment trap installations, precipitation, soil moisture and surface flow were measured continuously every 10 minutes at one plot of each land use type (m3, pawo2, paw2). Precipitation was measured with tipping buckets (ECRN-50 rain gauge, DecagonDevices), soil moisture was measured with a EC-5 sensor (DecagonDevices), and surface flow with a two-bowl tipping bucket, each bowl having 0.5 liter capacity (UP, 2006). The surface flow tipping bucket was installed at the outlet of the steel plate (Figure 2.2a). Water that flushed into the sediment trap was concentrated by the steel plate by means of the slightly v-shape plate and diverted to the tipping bucket. All data were logged by means of an Em50 Data Logger (DecagonDevices). To avoid underestimation of surface flow due to gaps between soil and geotextile, sediment traps were installed one year before data was used for evaluations. Thus, soil edges were fully regrown with grass when measurements started in spring 2007. At each of the nine sites 10 soil samples of the upper 5 centimeters were taken for grain size analyses. Also, grain size analyses were done for the eroded soil collected with sediment traps and sediment cups.

Sediment cups

Especially at the land use type pasture (with and without dwarf shrubs) the contribution area is heterogeneous because of frequent cattle trails crossing the plots. Thus, we installed sediment cups for the observation of small scale heterogeneity of soil erosion actions along each investigation plot (Figure 2.2a+c). The sediment cups do not have fixed boundaries. Thus, the collected data is not quantitative (as no contribution area is defined) but we rather aimed at getting qualitative information about small scale soil movement for each investigation plot. The sediment cups are modified after (Van Dijk et al., 2003). A sediment cup is composed of two layers. The outer layer consists of a robust material that is placed with the top flush at the soil surface. The flange is flattened and vee shaped for a smooth intersection between soil and cup and to avoid cavity (Figure 2.2b). The diameter at the top side is 9 cm. The inner layer is made of thin plastic and fits exactly into the outer cup. This inner layer can be easily replaced when the cups are sampled without disturbing the transition area of soil and cup. Both cups are permeable to water. The inner cup is perforated at the bottom. To avoid the loss of fine soil particles a filter paper (Sartorius Filter papers Type 3hw, filtration of particles >10 µm) was put on the bottom of the inner cup. 10 cups were placed at each investigation plot (Figure 2.2a) with a two meter interval. Soil erosion amounts that are measured with sediment cups were taken as bulk samples per each site. Thus, data for sediment cups that will be presented in this manuscript are shown for each of the nine sites with a mean value of ten cups per site.

2.3.3 Laboratory Cesium-137 measurement

For laboratory measurements soil samples of a depth of 10 cm were taken as over 70% of the Cesium-137 is stored in this section (Schaub et al., 2009). Each site was replicated five times (Figure 2.2a). Samples were collected during summer season 2007. Soil samples were dried at 40 °C, passed through a 2 mm sieve and finally ground using a WoC swing grinder. Soil samples were packed into 25 ml sample containers (Semadeni25) and measured for 8 hours. Measurements were done with a Li-drifted Ge-detector at the Department for Physics and Astronomy, University of Basel. In order to reduce the amount of radiation from background sources in the environment the samples were shielded by lead during measurement. Cesium-137 activity concentrations were determined using the InterWinner5 gamma spectroscopy software (Ortec). The resulting measurement error on Cesium-137 peak area is > 15%.

2.3.4 In-situ Cesium-137 measurement

A NaI scintillation detector with a 50.8 x 50.8 mm crystal was used for in-situ measurements in the Urseren Valley. For the measurement procedure, the NaI gamma spectrometer was placed perpendicular to the ground at a height of 25 cm and measured for 1h. Each site was replicated three times (Figure 2.2a). All measured Cesium-137 activities refer to 2007. Two reference sites near the valley bottom were measured with the same procedure. We define a reference site as a place which is neither influenced by erosion nor deposition. Deposition of eroded soil can be excluded because of lateral moraines between the steep slopes and the reference sites. Soil erosion on both reference sites was excluded due to the slope of both reference sites being 0% and a constant 100% vegetation cover since 1986. The latter can be confirmed from air photographs that were taken regularly since 1986. These reference sites were also taken for laboratory measurements that are described in chapter 2.3.5. To estimate the erosion rate from Cs-137 measurement we require the depth distribution of the Cs-137 concentration within the soil. The depth distribution was measured for a soil core taken at the reference sites. The soil core was portioned into slices of 2.5 cm thickness and the Cs-137 activity was measured in the laboratory with a GeLi-detector (Schaub et al., 2009). We found that the Cs-137 concentration decreases logarithmically with depth, whereby the concentration reduces by the half about every 5 cm. We derived the depth distribution of Cs-137 as:

$$Cs(z) = Cs(0)e^{-(-log(0.5)/b)z} \qquad 2.1$$

where z is the soil depth coordinate, *Cs(0)* the Cs-137 concentration in the uppermost layer, and b the distance in which the *Cs* concentration is bisected, i.e. in our case 5 cm. Knowing the activity at the surface and the shape of the Cs-137 depth distribution the value of *Cs(0)* can be found by summing up the gamma radiation of *Cs(z)* seen at the surface and equate it to the measurement of the NaI detector.

For detailed information on boundary conditions, measurement uncertainties including error propagation and the entire measuring procedure please see Schaub et al. (2009) and Konz et al. (2009b).

2.4 Results and Discussion

2.4.1 Erosion measurements with field installations

Monthly soil erosion rates measured with sediment traps during the vegetation periods 2007 and 2008 ranged between 0 kg ha^{-1} (hf2 and hf3 in September 2007) and 580 kg ha^{-1} (pawo2 in August 2007). The variation in collected sediment within single investigation plots is high with a low standard deviation of 0% for pasture with dwarf shrubs in October 2007 and high standard deviations up to 170% for pasture without dwarf shrubs in August 2007 and 173% for hayfields in September 2007 (Figure 2.3a). The high erosion rates as well as the high variation in collected sediment of hayfields (Figure 2.4) can be explained by soil activity due to mice just above the sediment trap of hf1 (measurements every second week resulted in lowest erosion rates of 4 g per plot (1 kg ha^{-1}) and highest erosion rates of 322 g per plot (80 kg ha^{-1})) during the vegetation periods. Hayfields 2 and 3 that are not affected by mice have low erosion rates and a low variation in collected sediment during the vegetation period ranging from 0 kg ha^{-1} to 8 kg ha^{-1} per month. The high variation in collected sediment of pawo (Figure 2.4) strongly correlates with soil detachment due to cow steps. The cows do not pass regularly the sediment traps. The eroded soil fragments at the sites pawo were up to 30 cm in size. This leads to the assumption that soil detachment was triggered by animal steps and that fragments were falling down due to gravity forcing.

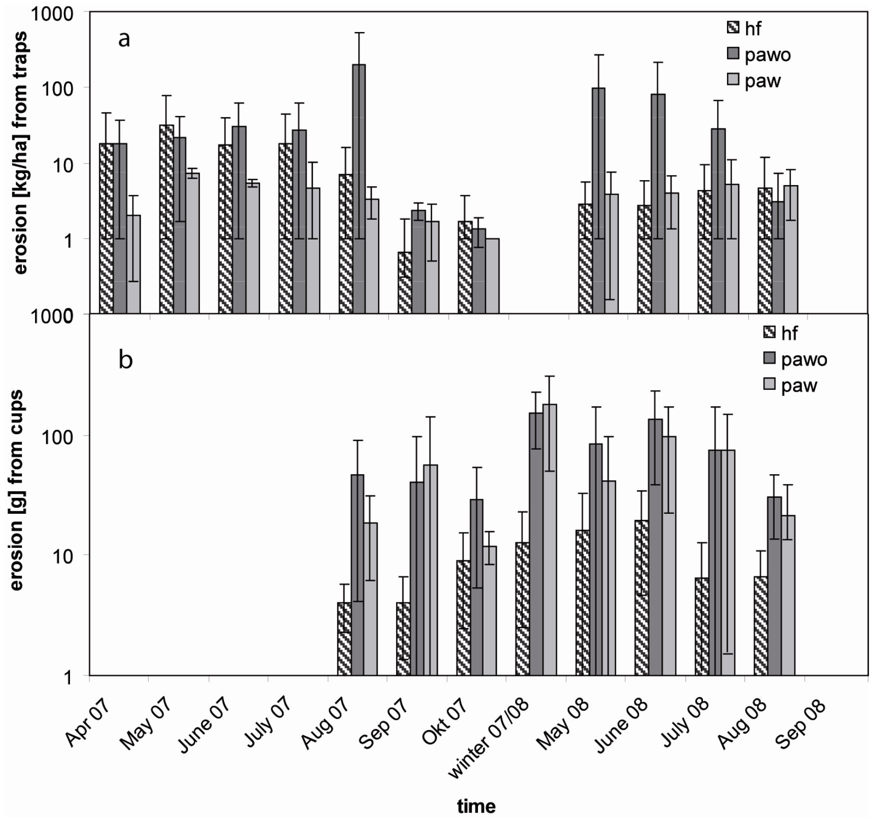

Figure 2.3: Monthly mean values of collected soil by sediment traps (upper figure) and sediment cups (lower figure) for the three land use types during the vegetation period 2007 and 2008.

Results of sediment traps indicate that hayfields and pastures with dwarf shrubs are less prone to erosion, whereas pasture without dwarf shrubs have higher damages. Pastures without dwarf shrubs are more prone to erosion than pastures with dwarf shrubs which might be due to the shielding effect of dwarf shrubs. Based on the measurement results with sediment traps erosion rates do not significantly ($p>0.05$) differ between single land use types (Figure 2.4).
Soil erosion rates from measurements with sediment traps correlate positively with slope steepness even though the range in slope was very small (Figure 2.5). No correlation was found between precipitation amount and erosion amounts ($p> 0.05$ by separating erosion values for rainfall events greater than 100mm per month and rainfall events smaller than 100mm per month) (Figure 2.6a) nor for overland flow and soil loss ($p>0.05$, data not shown). The overruling influence of soil detachment by animals (e.g. mice activity, sheep and cattle steps) most likely masked the influence of precipitation and surface flow amounts. This hypothesis is congruent with data from grain size analyses (Figure 2.7). Grain size distribution from eroded material at pastures without dwarf shrubs is within the heterogeneity of grain size analyses of the upper soil horizon (5 cm). Eroded material is partly in the form of whole soil pieces instead of single grains (e.g. silt). In contrast, eroded material from hayfields or from pastures with dwarf shrubs is finer (high percentage of smaller grain size

classes) compared to the bulk material of the upper horizon at the plots. Thus, we conclude that the triggering erosion process was not detachment by splash erosion and transport by water but rather detachment of whole aggregates and crumps by animal activity and transport by gravity forcing. If the hypotheses of detachment through animals would be true, we would conclude that meadows should significantly differ from measured erosion values on paw and pawo. By excluding hf1 where the activity of mice is great, meadows significantly differ from paw ($p=0.01$) and from pawo ($p=0.03$). The difference between measured erosion in dependence from land use type is also significant between paw and pawo ($p=0.04$).

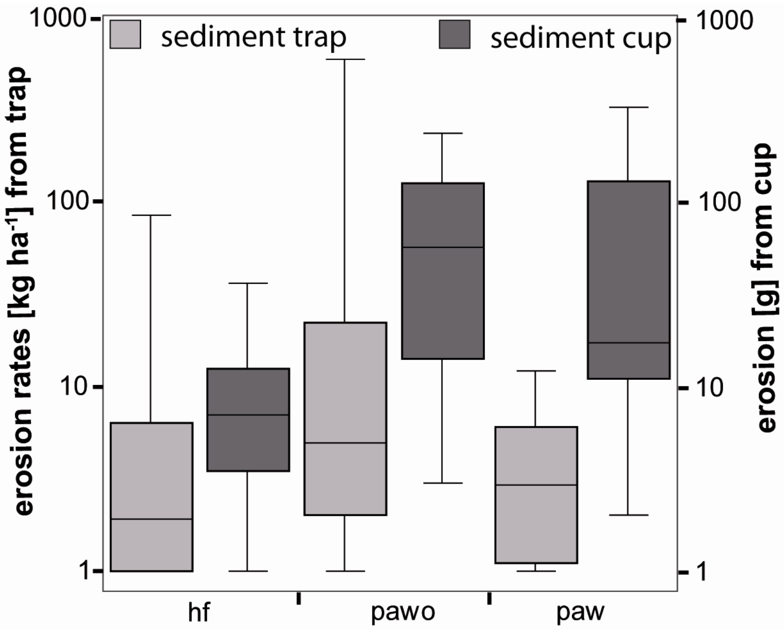

Figure 2.4: Monthly soil erosion measurements with sediment traps and sediment cups in dependence on the land use type.

Measured soil erosion rates are low and comparable to the measured erosion rates during the vegetation period from Felix and Johannes (1995) (0.1 to 200 kg ha^{-1}). Felix and Johannes (1995) conclude that their low erosion rates are based on low effective precipitation between 1 and 2%. The overland flow on our investigation plots is comparably low with a range of 0.6% and 2% from measured precipitation during the vegetation periods 2007 and 2008. Frankenberger (1995) measured much higher erosion rates during the vegetation period up to 20 t ha^{-1}. However, he reported that the effective precipitation during the investigation period was up to 60%.

Based on our measurement experience we suggest, that soil erosion measurement with sediment traps are generally very prone to errors. Since erosion events are subject to a vast spatial heterogeneity, it is critical to find the right location for the installation of a sediment trap. In contrast to soil erosion measurements on arable land, no defined ploughing rills are given that can be grasped by measurements. Thus, installation of sediment traps might just miss active erosion sites (Figure 2.8). A further problem of sediment traps is that they are not suitable for winter measurements because they will be flattened and destroyed by snow gliding (Figure 2.8b).

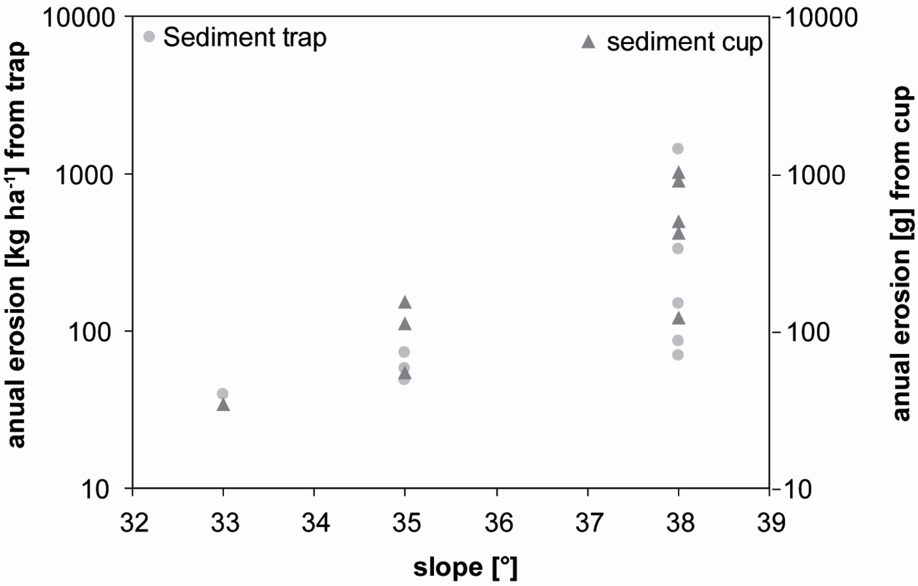

Figure 2.5: Measured soil erosion amounts are higher on steeper slopes. Low measured erosion rates occur independent of slope steepness on all sites. Thus there is no correlation between erosion rates and slope steepness for measurements with sediment traps and sediment cups (R^2 =0.5 for cups, 0.2 for traps) and the difference between measured erosion rates in dependence of slope steepness is not significant for both (p>0.05).

Since soil erosion rates during the vegetation periods were low, sediment cups were installed to test if there is a small scale soil movement within the test plots that is not captured because of soil particle sedimentation before they reach the sediment traps. Sediment cups have been installed right on the side of the sediment traps (Figure 2.2a+b). The absolute rates of the two measurement types can not be compared. With sediment traps is possible to install upper and side boundaries to refer erosion rates to a defined area. In contrast, sediment cups have to be installed without boundaries and have no defined catchment area (Figure 2.2a). Thus, the methods can only be compared qualitatively not quantitatively (e.g. high erosion activity or no erosion and relative differences between sites).

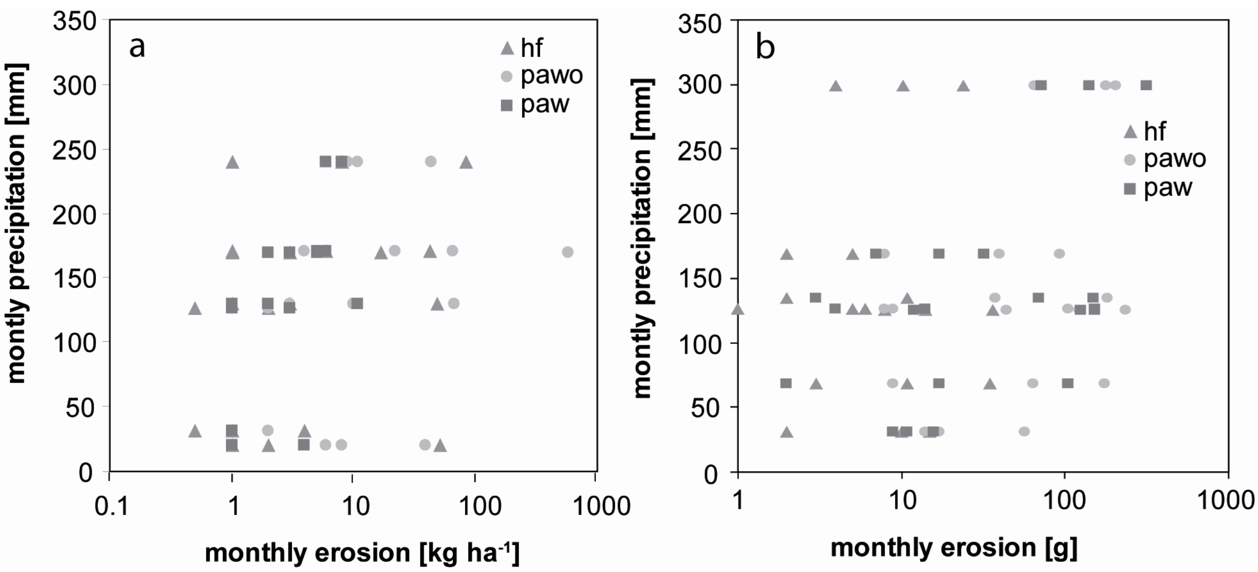

Figure 2.6: Correlation between monthly measured precipitation during 2007 and 2008 and the respective measured soil erosion with (a) sediment traps and (b) sediment cups. Correlated precipitation with erosion rates from sediment traps are for the vegetation periods 2007 and 2008. Correlation of precipitation and erosion from sediment cups include also precipitation during winter time 2007/2008.

Mean measured erosion rates of hf are 10 g per plot and month with a standard deviation of 97%. Mean measured erosion rates of pawo is 74 g per plot with an standard deviation of 98% and mean erosion values of paw was 78 g per plot with an standard deviation of 124%. Thus, the general pattern with low erosion rates in hayfields and high erosion rates in pastures without dwarf shrubs was congruent for sediment cups and sediment traps. However, sediment cups indicated a comparable high erosion activity for pasture with dwarf shrubs and for pastures without dwarf shrubs. The difference between the low erosion rates determined for pastures with dwarf shrubs with the sediment traps and the high erosion activity indicated by sediment cups point to small scale soil movement with detached particles not being transported down slope and thus not captured by sediment traps. Walling (1983) concluded that generally 70 – 85% of eroded material remains near the point of detachment. At our sites, the latter is obviously enhanced by the shielding effect of dwarf shrubs since results from sediment traps in pastures without dwarf shrubs show higher erosion rates than in pasture with dwarf shrubs while the small scale erosion activity indicated by sediment cups is comparable for both types (Figure 2.4). Like for the sediment traps, there was neither a correlation between collected soil and land use type nor precipitation amount and sampled soil in the cups (Figure 6b). The question arises how to judge the small scale erosion activity without transport of soil material down slope on pastures with dwarf shrubs. In a way this may enhance bioturbation of soil provided that detached soil is not transported down slope by winter processes which is obviously not the case at our sites (see Cesium-137 based erosion rates below).

Regarding the measurement procedure of soil erosion with sediment cups, this method turned out to be very suitable for alpine environments. The high small scale heterogeneity of soil detachment and deposition can be recorded and gives additional information to sediment traps. Furthermore, the installation of the cups is feasible even on steep segments and is connected to low efforts in cost and work time. During winter time, cups were not disturbed by snow packages or snow gliding processes. On most of the sites sediment traps were flattened whereas sediment caps were still in the right position. However, sediment cups have a limited volume thus underestimating erosion events with high material transport especially during the winter month. Snow cover and avalanche threat makes a regular sampling of cups impossible during the winter.

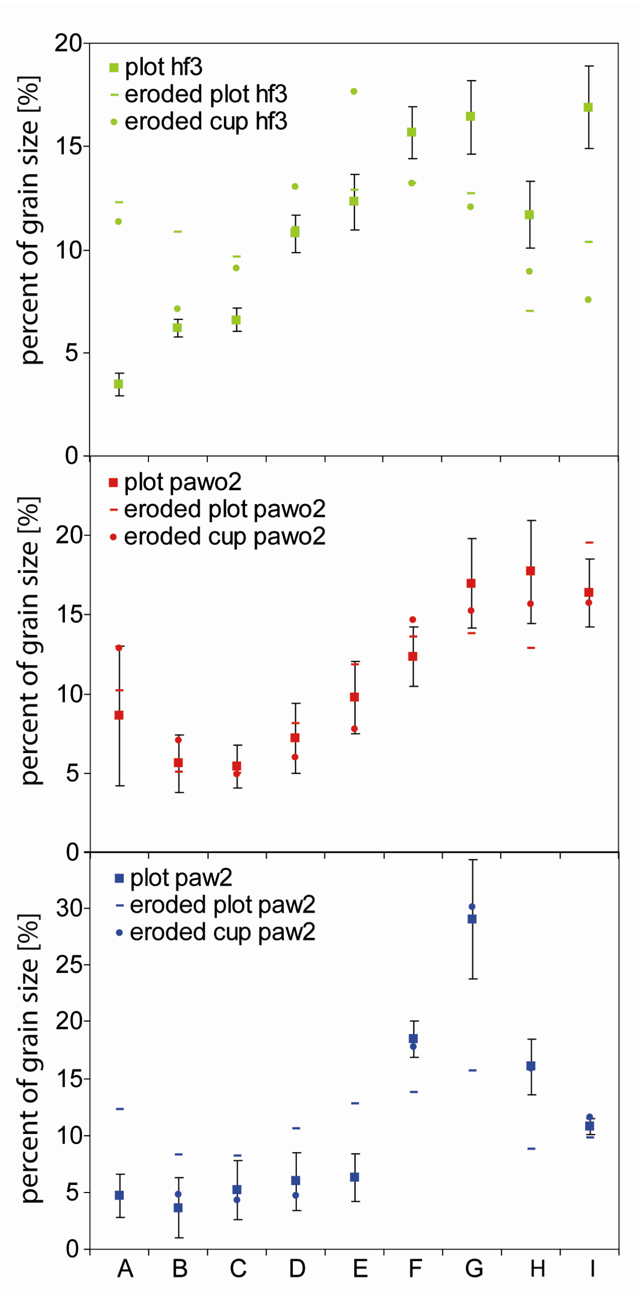

Figure 2.7: Grain size analyses of the upper 5 cm (plot) as well as eroded sediment that was collected with sediment traps (eroded plot) and sediment cups (eroded cup) for the sites hayfield 3, pasture without dwarf shrubs 2 and pasture with dwarf shrubs 2. A equals ≤2000 to >1000µm, B equals ≤1000 to >500µm, C equals ≤500 to >250µm, D equals ≤250 to >125µm, E equals ≤125 to >63µm, F equals ≤63 to ≥20µm, G equals 15 to ≥6µm, H equals 5 to ≥2µm and I equals 1.5 to <1µm.

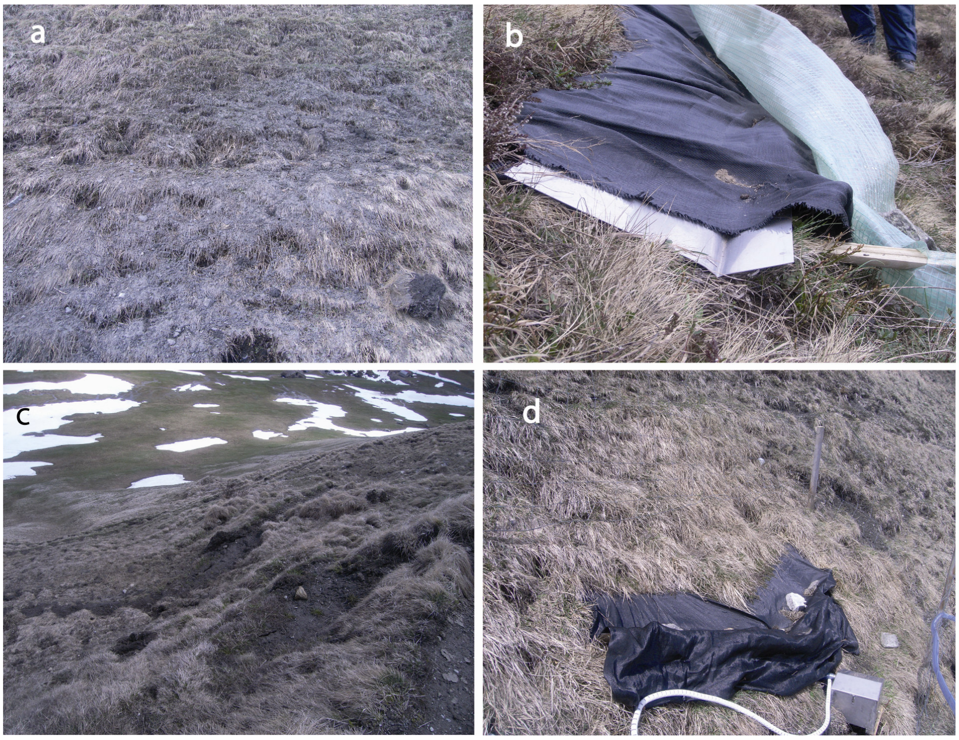

Figure 2.8: Detached soil on the ground after Winter 2007/2008 (upper figure, left side) and sediment trap on plot pawo3 without eroded material (lower figure and figure on right side), some meters away from detached material.

2.4.2 In-situ and laboratory measurements of Cesium-137

Erosion values based on laboratory analyses with a GeLi detector were consistent to erosion values from in-situ Cesium-137 measurements (Table 2.1). We have found a close correspondence ($R^2$of 0.94) between Cs-137 activities obtained from in-situ measurements (NaI-detector) and laboratory measurements (GeLi-detector) of several soil samples. This indicates that both methods are practicable for the quantification of soil erosion at alpine sites. However, the heterogeneity of soil erosion rates within single investigation plots is high with a maximum standard deviation of 151% for paw2 measured with the GeLi-detector (Table 2.1).

Regarding both Cs-137 measurement methods in-situ analyses turned out to be more suitable for alpine sites even if this method is not as established as measurements with the GeLi detector. Spatial distribution of Cesium-137 in grasslands is much more variable than in arable lands where Cesium-137 is mechanically homogenised by ploughing. Therefore, interpretation of laboratory Cesium-137 data of alpine grasslands is subject to errors relating to the small scale sampling. Our data shows that Cesium-137 varies largely over a small scale (Figure 2.10). Consequently, the number of soil samples per site must be increased to capture the variability and to achieve a representative Cesium-137 activity of a site. Collecting big amounts of samples in the field for laboratory analyses does not suitable for sensitive mountain soils seriously affected by soil erosion. Especially in regions where soil recovery takes hundreds of years, non-destructive in-situ measurements should be favoured. In-situ measurements integrate over 8 m^2 and averages small scale variability. In addition, the method is non-destructive (Konz et al., 2009a). The disadvantage of the in-situ measurements with the NaI-detector is an uncertainty of about 17% due to manual analyses of spectra. Small changes in start and end position of the peak leads to a big variability in peak area (Konz et al., 2009b). This error on peak area was determined by using the mean standard deviation of peak areas of 20 test spectra evaluated by five persons independently. The source area of the spectra in the field was unknown to avoid manipulation of evaluation.

Table 2.1: Cesium-137 based soil erosion amounts for sites measured with NaI- and GeLi- detector. Additionally for the NaI based erosion values, mean annual height for soil loss is given as well as soil density for each site. Values in the square brackets are the mean standard deviation due to the heterogeneity on each plot based on three measurements with the NaI- detector and five measurements with the GeLi detector. Fractional vegetation cover was measured in April (first value) and September (second value) 2007 (s.d. 5%, n=3).

land use type	soil density [kg m^{-3}]	fractional vegetation cover [%]	erosion from NaI-detector [t ha^{-1} a^{-1}]	erosion from GeLi detector [t ha^{-1} a^{-1}]
hf1	1066.2	77/92	35.5 (± 7.2)	28.8 (± 4.6)
hf2	1043.8	93/95	13.5 (± 2.3)	12.3 (± 8.8)
hf3	1041.8	76/90	34.4 (± 6.4)]	27.6 (± 12.5)
pawo1	1357.4	65/65	17.4 (± 5.0)	17.2 (± 13.2)
pawo2	1336.3	62/62	20.8 (± 9.4)	19.3 (± 19.3)
pawo3	1242.4	67/67	27.8 (± 11.3)	24.6 (± 9.7)
paw1	1470.5	77/77	8.0 (± 2.8)	8.3 (± 12.2)
paw2	1165.0	79/79	7.8 (± 2.0)	6.7 (± 12.6)
paw3	1028.5	73/73	14.5 (± 3.1)	17.9 (± 10.9)

The reference sites in the Urseren Valley had a mean Cesium-137 activity of 146.4 Bq kg^{-1} (s.d. ± 17.3%) measured by GeLi detector in the laboratory. Resulting erosion rates are between 6 and 37 t ha^{-1} a^{-1} (Figure 2.9). Mean Cesium-137 activity based on in-situ measurements was 91 Bq kg^{-1} (s.d. ± 19.2%) for all hayfields, 94 Bq kg^{-1} (s.d. ± 27.4%) for all pastures without dwarf shrubs and 121 Bq kg^{-1} (s.d. ± 27.3%) for all pastures with dwarf shrubs. The variation in erosion rates for the three measured hayfields is high. This variation might be explained by avalanche impact on hayfield hf3 (Ambuehl, 1961) and mice activity (see chapter 2.4.1) combined with snow gliding processes on hayfield hf1. High avalanche frequency might cause higher erosion rates during winter time and early spring. In contrast, at hayfield hf2 with low erosion rates neither high activity of avalanches nor snow gliding nor mice activity was observed. Hayfield hf2 and pasture with dwarf shrubs paw1 had similar erosion rates even though the fractional vegetation cover was significantly lower in pasture paw1 (77%) than in hayfield hf2 (92%).

The latter might be due to increased sedimentation induced by dwarf shrubs which stabilize the soil and act as physical barriers, thus, reducing transport of soil particles down slope. That dwarf shrubs might act as physical barriers was also noticed during the vegetation periods 2007 and 2008 with sediment cup measurements (see above). The variation in Cesium-137 based erosion rates within single sites is higher on pastures than on hayfields (Figure 2.9). This might be based on the high micro morphology on the sites that leads to a high heterogeneity within a small scale. This micro morphology is due to cattle trails and a high skeleton content that influences in-situ measurements of Cesium-137.

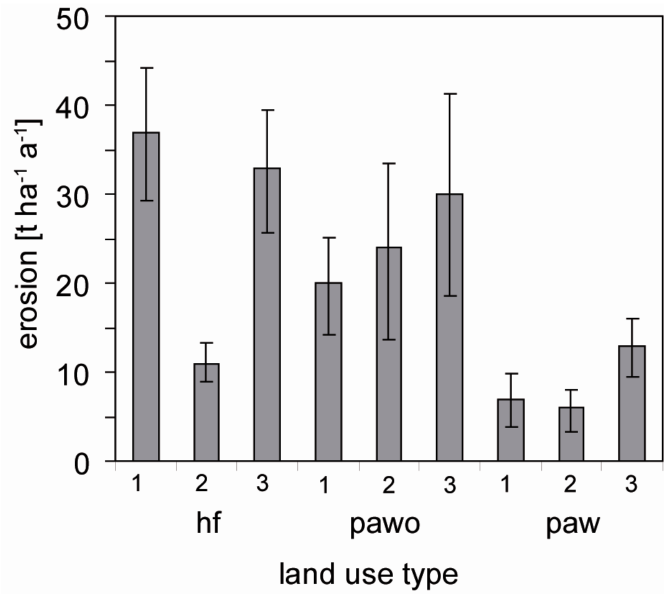

Figure 2.9: Erosion rates from in-situ Cesium-137. Error bars are due to manually analysis of gamma spectra as well as plot heterogeneity.

Generally, erosion values based on sediment traps are significantly lower by two orders of magnitude than erosion values based on Cesium-137 measurements. The latter integrate erosion over a time period from April 1986 to today. Thus, erosive processes during winter time and snow melt as well as during intensive rain storm events are included. There was no intensive rainfall event during the investigation period 2007 and 2008 (data not shown). The most intensive rainfall events since 1986 occurred in 1987, 1991 and 2002 during the vegetation periods.

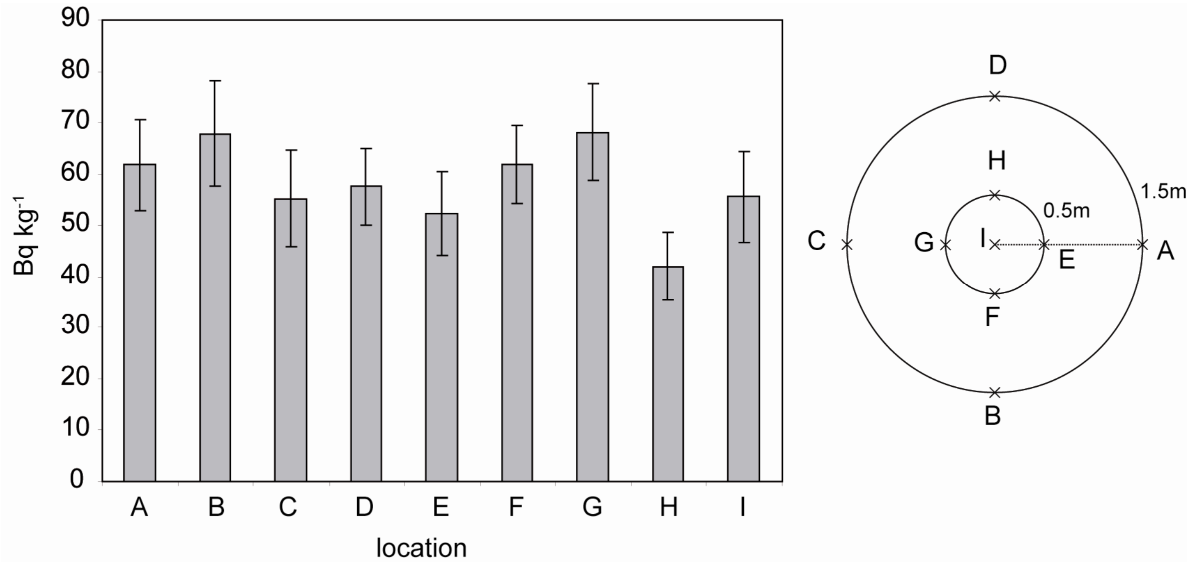

Figure 2.10. Laboratory Cesium-137 measurements for single soil probes (plan of soil sampling on the right hand side) on the land use type hf3 were an intensive soil sampling was done to show the large heterogeneity of alpine sites.

To separate erosive rainfall events during the vegetation period from 1986 till 2008 the WEPP model was applied for this period (Konz et al., 2009a). Model results indicated that erosion rates during all vegetation periods were low even if heavy rainfall events occurred and that over 98% of the soil was eroded during October and April and not during the vegetation period. The latter might be due to the very high infiltration capacity of the investigated soils (Merz et al. 2009). Thus, erosion processes during the vegetation period seems to have a minor influence on annual soil erosion rates in the Urseren Valley. Erosion processes during the vegetation period are most likely of minor importance if the effective rainfall is < 2%. This situation can be found in the Urseren Valley (this study) as well as in the investigation area Jenner near Berchtesgarden published by Felix & Johannes (1995). However, Frankenberger et al. (1995) described that with an effective rainfall up to 60% (Allgäu, south-east Germany) erosion rates of > 20 t ha^{-1} can occur during the vegetation period. The latter might have important implications if we consider climate change scenarios: an increase in torrential rain events might significantly increase soil erosion during the vegetation period.

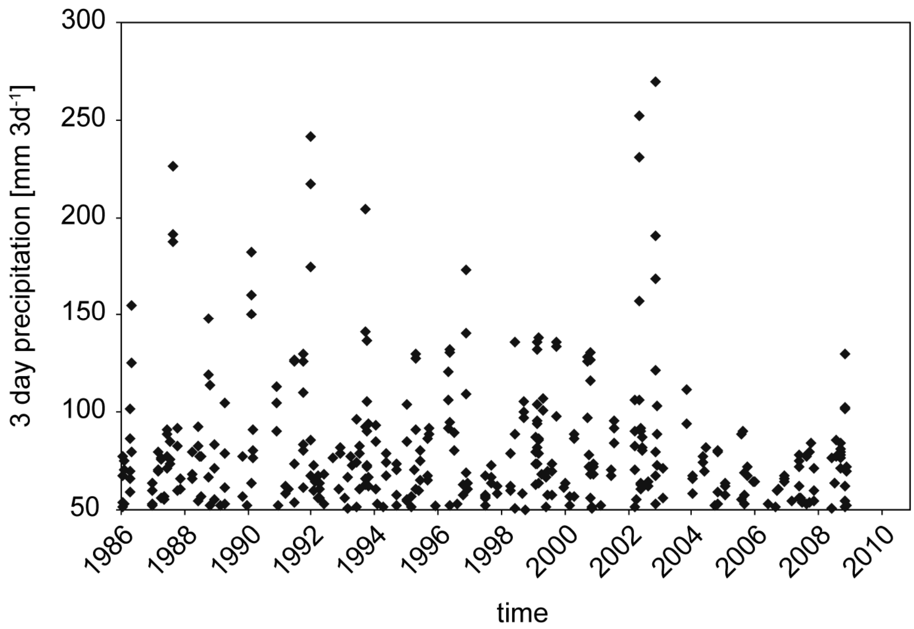

Figure 2.11: Precipitation greater than 50 mm per 3 days from 1986 till 2008 for the Urseren Valley.

Table 2.2: Advantages and disadvantages of applied measurement methods in the Urseren valley for soil erosion quantification in alpine areas with snow influence during winter time
Measurement.

measurement method	advantage	disadvantage
sediment trap	• defined area • Surface flow measurement can be included	• destruction during winter time possible • finding the right position should be based on further experience • temporal resolution depends on time effort • destructive (installation) and time intensive method • finding the right position has to be based on subjective expert knowledge
sediment cup	• applicable at variable land use (also ski piste) • high replicates possible • measurements during winter possible • easy installation	• temporal resolution depends on time effort • limited erosion amount measureable • automation is hardly possible • no quantification of erosion rates (qualitative)
Cs-137 in-situ (based on NaI detector measuerements)	• applicable at variable land use (also ski piste) • summer and winter processes are included • non destructive method • short measurement time (1h)	• measurement uncertainty is high • high temporal resolution not possible • differentiation between eroded sediment from different seasons not automatically possible
Cs-137 Laboratory (based on GeLi detector measuerements)	• high peak resolution • applicable at variable land use (also ski piste) • summer and winter processes are included • measurement uncertainty is lower than in.situ measurements	• destructive method (soil sampling) • long measurement time per soil sample (8h) • high temporal resolution not possible • expensive (equipment, soil preparation) • differentiation between eroded sediment from different seasons not automatically possible

2.5 Conclusions and Perspectives

Erosion rates in an alpine environment between 1550 and 1800 m a.s.l were determined in a measurement comparison approach. We distinguished between measurements with sediment traps applicable during the vegetation period and Cs-137 based erosion measurements which integrate since 1986 and thus include long term soil erosion due to extreme and/ or winter events. Regarding a comparison of applied erosion measurements all measurement has inherent advantages as well as disadvantages on alpine sites (Table 2.2). The main conclusion here might be that the measurement method has to be adapted to the main research question. Even though the latter might be considered a trivial statement, it is more than crucial in alpine environments and was often been neglected in past studies.
Both Cs-137 methods (laboratory measurements with a GeLi detector based on soil sampling as well as in-situ measurements with a portable NaI detector) yielded consistent erosion rates (R^2 = 0.94). Because alpine sites are very heterogeneous at a micro scale (<1 m^2) a high number of soil samples per site are required for laboratory analyses. Collecting numerous samples in the field does not seem suitable for mountain grassland soils seriously affected by

soil erosion. Especially in regions where soil recovery takes hundreds of years, non-destructive in-situ measurements should be favoured.
The Cs-137 based erosion rate measurements were significantly higher ($p<0.05$) than erosion rates based on sediment traps during the vegetation period. We conclude that this difference is due to extreme events and winter processes captured by the Cs-137 measurements. Most likely the influence of snow including snow gliding, avalanches as well as increased overland flow during snow melt is the major driver for soil erosion in alpine grassland systems.
With the effective precipitation being only about 2%, erosion rates due to water erosion during the vegetation period were low for all investigated alpine grassland sites. Erosion induced directly by animal activity (cattle or sheep steps, mice digging) was a bit higher but still negligible compared to Cs-137 based erosion rates. However, it is very likely that animal activity increases sensitivity of sites leaving them vulnerable for winter processes. Nevertheless, two out of three investigated hayfields had surprisingly high Cs-137 based erosion rates comparable to pastures. The latter might be a problem of low sample numbers with one hayfield being a site affected by high mice activity and another by high avalanche frequency.
Sediment cups are a practicable tool to determine small scale erosion activity. Without upper and side boundaries, the method is qualitative. Sediment cups are also useful to detect erosion activity during the winter time. None of our 90 cups that have been installed were destroyed during the winter. Our work provides a further step of soil erosion knowledge in alpine grasslands. To develop soil erosion prediction models in the future winter processes as driving factors of soil erosion have to be investigated quantitatively and qualitatively and more data are needed for the calibration and validation of these models.

CHAPTER 3

Cs-137 based erosion rate determination of a steep mountainous region

Accepted by: Journal of Plant Nutrition and Soil Science,
Vol. 172, Issue 5, pp.615-622, 2009
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3.1 Abstract

Data on quantification of erosion rates in alpine grasslands remain scarce but are urgently needed to estimate soil degradation. We determined soil erosion rates based on Cs-137 in-situ measurements. The method integrates soil erosion over the last 22 years (time after the Chernobyl accident). Measured erosion rates were compared with erosion rates modeled with the Universal Soil Loss Equation (USLE). The comparison was done in order to find out if the USLE is a useful tool for erosion prediction in steep mountainous grassland systems. Three different land use types were investigated: hayfields, pasture with dwarf shrubs and pasture without dwarf shrubs. Our test plots are situated in the Urseren Valley (Central Switzerland) with a mean slope steepness of 37°. Mean annual soil erosion rates determined with Cs-137 of the investigated sites ranged between the minimum of 4.7 t ha^{-1} a^{-1} for pastures with dwarf shrubs to >30 t ha^{-1} a^{-1} at hayfields and pastures without dwarf shrubs. The determined erosion rates are 10 to 20 times higher compared to previous measurements in alpine regions. Our measurements integrated over the last 22 years, including extreme rainfall events as well as winter processes, whereas previous studies mostly reported erosion rates based on summer time and short term rainfall simulation experiments. These results lead to the assumption that heavy rainfall events as well as erosion processes during winter time and early spring do have a considerable influence on the high erosion amounts that were measured. The latter can be confirmed by photographs of damaged plots after snow melt. Erosion rates based on the USLE are in the same order of magnitude compared to Cs-137 based results for the land use type pasture with dwarf shrubs. However, erosion amounts on hayfields and pasture without dwarf shrubs are underestimated by the USLE compared to Cs-137 based erosion rates. We assume that the underestimation is due to winter processes that causing soil erosion on sites without dwarf shrubs that is not considered by the USLE. Dwarf shrubs may possibly prevent from damage of soil erosion through winter processes. The USLE is not able to perform well on the affected sites. Thus, a first attempt was done to create an alpine factor for the USLE based on the measured data.

3.2 Introduction

Soil erosion in the Alps is a well recognized problem, identified as a priority for action within the EU (European Union) soil protocol to the Alpine Convention (2005). Steep slopes, extreme climate, fragile soils and partly intensive agricultural land use characterize the Alpine environment. Main drivers for soil erosion are topography, soil texture, land cover, rainfall patterns and land use. Additionally, snow processes like snow melt or avalanches enhance erosion in alpine regions. A comprehensive assessment of soil erosion on the Alps is still missing (ClimChAlps, 2006). Mapping and quantification of soil erosion under different land use conditions has been studied in numerous projects for agricultural soils in lowlands or low mountain ranges (e.g. Gabriels et al., 2003; Leser et al., 2002; Nearing et al., 1999; Prasuhn et al., 2007) as well as in laboratory experiments (e.g. Boubakari and Morgan, 1999). However, only few studies exist on soil erosion measurement and quantification under natural precipitation regimes in alpine environments (Descroix and Mathys, 2003; Felix and Johannes, 1995; Isselin-Nondedeu and Bedecarrats, 2007). The latter studies do all focus on soil erosion measurement during the vegetation period. Hence, more data on soil erosion in alpine regions for longer time periods including also winter processes is needed.

Numerous models have been developed and used to quantify and predict soil erosion. The Universal soil loss equation USLE (Wischmeier and Smith, 1978a) is a frequently used model to identify erosion risk areas and to help in decision making processes for long term average erosion development in agricultural lowlands (e.g. Auerswald et al., 2003). Erosion rates are calculated as

$$A = R\,K\,L\,S\,P\,C \qquad 3.1$$

where A is the average annual soil loss (kg m^{-2} a^{-1}) due to erosion, R is the rainfall-runoff factor (N h^{-1}), K is the soil erodibility factor and gives the soil's tendency to erode (N^{-1} h kg m^{-2}). The topography factor combines slope length L (-) and slope steepness S (-), P is the support practice factor (-) and C is the cover and management factor (-). C factors are well defined for crop rotations (e.g. Gabriels et al., 2003). Wischmeier and Smith (1978a) suggest a C factor of 4 10^{-3} for established hayfields with a mean annual hay production of 6.7 t ha^{-1} and a fractional vegetation cover of >95 %, and 3 10^{-3} for permanent pastures. The values are increasing with decreasing fractional vegetation cover. Prasuhn et al. (2007) used a C factor of 3 10^{-3} for woodland and grassland for the Swiss erosion hazard map.

The determination of long-term soil erosion rates and patterns of soil redistribution on cultivated land via Cs-137 is an established method (e.g. He and Walling, 2000; Ritchie and McHenry, 1990). The use of Cs-137 measurements to quantify soil redistribution rates is commonly based upon a comparison of Cs-137 inventories for individual sampling points to the local reference inventory. When Cs-137 reaches the soil surface by wet deposition, it is tightly adsorbed to fine soil particles (clay minerals and organic matter). The subsequent lateral redistribution of adsorbed Cs-137 is associated with soil erosion (Sawhney, 1972). Erosion is indicated by lower Cs-137 values, while sedimentation is indicated by higher Cs-137 inventories compared to the reference site (e.g. Walling and He, 1999).

The aim of this study was to determine long term all-season soil erosion rates for different land use conditions in an alpine catchment of Switzerland. Large damages are visible on the investigated sites. The knowledge of the quantification of soil erosion is urgently needed in the Swiss Alps to address the need of soil conservation strategies for the future. Additionally, since no erosion prediction model for steep alpine environments is available, we tested the USLE as it is one of the most used erosion models worldwide. Cs-137 based erosion rates were compared to modeled results of the USLE (A) in order to assess the suitability of the model for steep alpine grasslands. The Cs-137 amounts were determined with in-situ NaI (Sodium-Iodide) gamma detector measurements. The advantages of such in-situ measurements in alpine areas are discussed by Schaub et al. (2009).

3.2 Material and Methods

3.2.1 Study site

The study area is located in Central Switzerland (Canton Uri) in the Urseren Valley (Figure 3.1). The elevation of the W-E extended mountain valley ranges from 1450 to 3200 m a.s.l. Mean annual rainfall from 1986 to 2007 was 1516 mm and mean air temperature was 4.3°C at the valley bottom. The valley is typically snow covered from November to April with the maximum snow height in March (Ambuehl, 1961) and a mean annual snowfall between 1986 and 2008 of 448 mm (Source: MeteoSwiss, 2007). Runoff is usually dominated by snowmelt in May and June. Land use in the valley is hayfield near the valley bottom and pasture further upslope. Siliceous material is dominant, forming cambic podzols (anthric) and podzols (anthric) classified after IUSS Working Group (2006). The characteristic of these soils is a migration (M-horizon) horizon within the upper 100 cm that is caused by sedimentation in the past. The thickness of the M- horizon is between 5 and 45 cm. For a detailed description of the Urseren Valley see Meusburger and Alewell (2008).
All investigated plots are situated on the south-facing slopes between 1550 and 1800 m a.s.l. (Figure 3-1). Three different grassland types were investigated with three replicates: hayfield (hf), pasture with dwarf shrubs (paw) and pasture without dwarf shrubs (pawo). The slope steepness of the investigation plots is between 35°-39°. Grain size analyses of all investigated plots are listed in Table 3.1. Dominant plant species in the hayfields are Trifolium pratense ssp. Partense, Festuca sd, Thymus serpyllum and Agrostis capillaries. Pasture with dwarf shrubs are dominated by Calluna vullgaris, Vaccinium myrtillus, Festuca violacea, Agrostis capillaries and Thymus serpyllum. Main species of pastures without dwarf shrubs are Glubelaria cordifolia, Festuca sd and Thymus serpyllum.

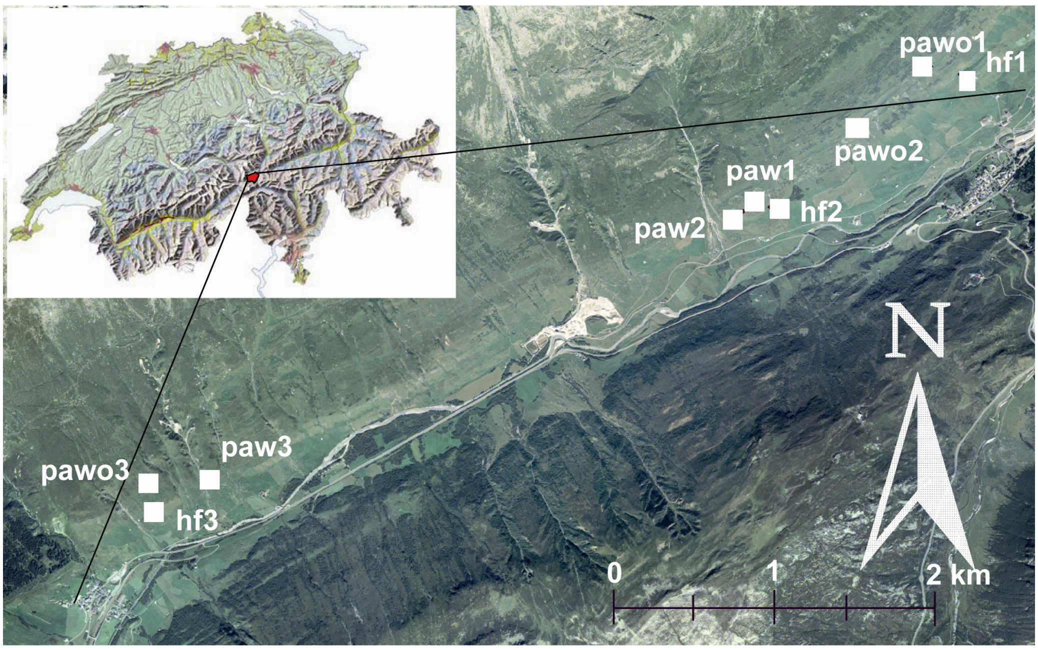

Figure 3.1: The Urseren Valley in southern Switzerland and the location of the investigated sites with three grassland types: hayfield (hf), pasture without dwarf shrubs (pawo) and pasture with dwarf shrubs (paw).

3.2.2 Erosion assessment with Cs-137

We used a NaI scintillation detector with a 50.8 x 50.8 mm crystal to measure Cs-137 distribution in the field. For the one hour measurements the gamma spectrometer was placed perpendicular to the ground at a height of 25 cm. The Cs-137 measurements were done in summer and autumn 2007 with three replicates at each plot. Additionally, two reference sites near the valley bottom which are unaffected by erosion were measured with the same procedure.

We define a reference site as a place which is influenced neither by erosion nor deposition. Both reference sites are located at the valley bottom. Deposition of eroded soil on the reference sites can be excluded because of lateral moraines between the steep slopes and the reference sites. Soil erosion on both reference sites is unlikely due to a constant 100 % vegetation cover since 1986. This can be confirmed from air photographs that were taken regularly since 1986. Additionally, the slope of both reference sites is 0 %.

Table 3.1: Measured and calculated parameters for the investigation sites in the Urseren Valley. Relative standard deviation (*n = 10 soil samples) of grain size analyses is 10 %. Relative standard deviation of organic matter (n = 10 soil samples) is 17 %. K is given in N h kg m-2 and R in N^{-1} h^{-1}. (**vfs = very fine sand and is required in the k-nomograph of Wischmeier and Smith (1978)).

	grassland type								
	hayfield			pasture without dwarf shrubs			pasture with dwarf shrubs		
	hf1	hf2	hf3	pawo1	pawo2	pawo3	paw1	paw2	paw3
slope [°]	39 (±1)	38 (±1)	35 (±1)	38 (±1)	38 (±1)	35 (±1)	38 (±1)	38 (±1)	35 (±1)
*sand [%]	40.2	23.8	39.3	37.9	36.6	39.4	49.8	25.7	44.9
*silt [%]	47.3	58.8	43.8	50.5	47.1	45.6	37.6	63.5	40.9
*clay [%]	12.5	17.3	16.9	11.6	16.4	15	12.5	10.8	14.2
silt & **vfs	60.2	68.5	56.1	61.7	56.8	58.0	48.5	69.8	52.1
org. mat. [%]	13.2	12.3	12.7	13.1	12.8	12.4	11.7	11.9	12.2
pH value	5.0	4.4	4.5	7.1	7.3	4.6	4.3	4.4	4.5
R	97.2	94.5	93.6	97.6	96.4	96.4	94.1	91.7	94.8
K	0.28	0.29	0.23	0.27	0.23	0.25	0.21	0.32	0.23
P [-]	1	1	1	0.9	0.9	0.9	0.9	0.9	0.9
S [-]	10.1	9.8	9.0	9.8	9.8	8.5	9.8	9.7	9.1
L [-]	2.2	0.9	2.3	1.2	1.4	1.8	1.4	1.3	1.3
C [-]	1×10^{-2}	6×10^{-3}	1×10^{-2}	2×10^{-2}	3×10^{-2}	2×10^{-2}	4×10^{-2}	4×10^{-2}	4×10^{-2}

To estimate the erosion rate from Cs-137 measurement we require the depth distribution of the Cs-137 concentration within the soil. The depth distribution was measured for a soil core taken at the reference sites. The soil core was portioned into slices of 2.5 cm thickness and the Cs-137 activity was measured in the laboratory with a GeLi-detector (Schaub et al., 2009). We found that the Cs-137 concentration decreases logarithmically with depth, whereby the concentration reduces by the half about every 5 cm. We derived the depth distribution of Cs-137 as:

$$Cs(z) = Cs(0)e^{-(-log(0.5)/b)\, z} \qquad 3.2$$

where z is the soil depth coordinate, $Cs(0)$ the Cs-137 concentration in the uppermost layer, and b the distance in which the Cs concentration is bisected, i.e. in our case 5 cm. Knowing

the activity at the surface and the shape of the Cs-137 depth distribution the value of *Cs(0)* can be found by summing up the gamma radiation of *Cs(z)* seen at the surface and equate it to the measurement of the NaI detector.
To determine in-situ erosion rates the activity at the test plot is measured with the NaI detector. Soil density was measured tree times for the first 10 cm with the gravimetric method. Soil samples with a defined volume were taken dried for 3 days at 105°C in the oven and weighted (Table 3.2). To evaluate this activity we make the assumption that the Cs-137 depth distribution was the same as at the reference site at Cs-137 deposition 1986 and that no new soil material is deposited on the test plots. Under the assumptions that these conditions are fulfilled we first correct the reference Cs-137 depth profile according to the half life time of Cs-137, i.e. the time step of the reference measurement and the test plot measurement have to be known. The erosion rate can be calculated by removing a soil layer from top of the Cs-137 profile. The thickness of the removed soil layer must have such a thickness that the sum of the gamma radiation of *Cs(z)* seen at the surface and equates to the measurement of the NaI detector at the test plot. Having the thickness of this layer and knowing the time when Cs-137 was deposited assuming continuous erosion processes, the annual erosion rate can be determined.
Since the Urseren Valley is dominated by steep hill slopes it is very likely that the upper most soil layer gets mixed (especially at the land use types pasture), e.g. due to solifluction and bioturbation. This fact is underlined by the frequent occurrence of migration horizons (section 3.2.1). Thus, we extended our analysis by implementing a mixture horizon, i.e. we define an uppermost possible layer thickness (1 – 5 cm) for which we continuously calculate the mean of the Cs-137 concentration. By applying this technique mixing processes within the topsoil can be considered to get a better estimation of erosion rates and their uncertainty due to upper mixing layers.

The error of the erosion value of each single investigation plot (Figure 3.4) originates from different sources. Around 10 % (depending on the land use type) of the uncertainty of the Cs-137 based erosion values is based on the heterogeneity of Cs-137 values gained by the replicated measurements on each single plot. The heterogeneity is higher on the pastures and lower on the hayfields. Another error source is due to the conversion from Cs-137 measurements to erosion data since a mixture depth of the upper 5 cm has to be expected. Finally, the measurement error due to the analyses of the of Cs-137 spectra is 17.3 % since spectra have to be analyzed manually. Small changes in start and end position of the peak leads to a big variability in peak area. This error on peak area was determined by using the mean standard deviation of peak areas of 20 test spectra evaluated by five persons independently. Spectra have been analyzed using a VisualBasic program provided by H. Surbeck, University of Neuchâtel. This program allows for setting regions of interest (ROI) and determining gross and net ROI areas. Background within the ROI is assumed to be a linear function between averaged background values at the left and the right ROI margins, respectively. The Region of interest was set to include the three interfering peaks Tl-208 at 583.2 keV, Be-214 at 609.3 keV and Cs-137 at 661.7 keV. The Peak area of Cs-137 is specified by a gauss function over the region of interest (ROI). The total erosion estimated by Cs-137 measurements includes the cumulative erosion since 1986 (time after deposition of radioisotopes due to the Chernobyl reactor accident). Cs-137 input from atomic bomb tests in the early sixties had minor impacts on the Urseren Valley (Schaub et al., 2009). Therefore, a mean annual soil erosion rate can be estimated for the time period 1986-2007.

Table 3 2: Cs-137 activity, soil density and resulting erosion rates. The Cs-137 reference value is about 145 Bq kg^{-1}. Soil density is determined for the first 10 cm (*n = 3 soil samples per site).

	grassland type								
	hayfield			pasture without dwarf shrubs (pawo)			pasture with dwarf shrubs (paw)		
	hf1	hf2	hf3	pawo 1	pawo 2	pawo 3	paw 1	paw 2	paw 3
Cs-137 [Bq kg^{-1}] (RSD %)	73.9 (1.3)	118.6 (2.4)	79.8 (11.1)	102 (11.9)	95.3 (25.6)	84.1 (18.0)	128.3 (7.2)	131.7 (15.0)	115.8 (13.5)
*soil density [kg m^{-3}] (RSD %)	1066 (2.1)	1044 (2.6)	1041 (1.9)	1357 (3.4)	1336 (2.3)	1242 (4.1)	1270 (5.7)	1165 (3.8)	1026 (2.9)
erosion t ha^{-1} a^{-1}	37	11	33	20	24	30	7	6	13

3.2.3 Erosion assessment with the USLE

The USLE was originally parameterized for soil erosion evaluation in the United States. Modified versions for other regions as well as for different temporal resolutions have been developed in the past. These models are for example the German USLE that is called ABAG and the revised version of the USLE, the RUSLE. For our study, single factors of these tree models were chosen depending on the best suitability for our site. The factors were either determination from field data or taken from literature values. Thus, we did not only take the factors as proposed by Wischmeier and Smith (1978) for the USLE but tried to get the most appropriate factor for our land use site since no special model is designed for mountainous regions. We used the algorithm of Rogler and Schwertmann (1981) (ABAG) to determine R since the equation was designed for a comparable geographical and meteorological situation. R was calculated from daily precipitation data of the local meteorological station of MeteoSwiss (2007) (Tab. 3.1). In the Alps it is necessary to consider that some of the precipitation falls as snow, which reduces R factor values. Schuepp (1975) found the following relation between the elevation (m a.s.l.) and proportion of snow (f(x)):

$$f(x) = 0.0264\,h - 2.0663 \qquad 3.3$$

The combined equation for the *R*-factor is:

$$R = 0.083\,N\,(1-((0.0264\,h - 2.0663)/100) - 1.77 \qquad 3.4$$

where *R* is the rainfall-runoff factor (N h^{-1}) and *N* the average annual rainfall (mm). The elevation h (m a.s.l.) of the location is implemented in the equation with the result of decreasing rainfall amount and increasing snow amount with increasing height (eq. 3.3). The soil erodibility factor *K* (N^{-1} h kg m^{-2}) was calculated according to the *K* nomograph after Wischmeier and Smith (1978) (USLE).

Information for the *K* factor was calculated for the top 10 cm using grain size analyses and content of organic matter. Total carbon content of soils was measured with a Leco CHN analyzer 1000 and grain size analyses were measured with sieves for grain sizes between 32-1000 µm and with a Sedigraph 5100 (Micromeritics) for grain sizes between 1-32 µm.

L and *S* are topographic factors typically combined into *LS* factor. *L* scales the distance of upslope flow accumulation at any field location to the length of the standard USLE plot used for experimental erosion measurements (22.13 m). S scales the steepness at the field site relative to the slope of the experimental USLE plots (9 %; 5.1°) and was calculated after Renard et al. (1997) (RUSLE) for this investigation as it is valid also for steep slopes.

$$L = (\lambda / 22.13)^{m} \qquad 3.5$$

with $m = \beta/(1+\beta)$

$\beta = (\sin\theta/0.0896)/(3 \times (\sin\theta)^{0.8} + 0.56$

$$\text{for } \theta\ 5.14° \qquad S = 16.8 \times \sin\theta - 0.5 \qquad 3.6$$

where L is the slope length factor (-), λ is the real slope length (m), m the slope length exponent (-), β the susceptibility to rill erosion (-), θ slope gradient (°) and S the slope factor (-).
The support and practice factor P (-) was set to 0.9 after Wischmeier and Smith (1978) (USLE) for all pastured plots (Tabel 3.1) since alpine pastures have small terrace buildings in the form of cattle trails on these land use types which are suggested to be considered in P (Foster and Highfill, 1983; Wischmeier and Smith, 1978). On hayfields P was set to 1 since not terraces do exist. Although terrace building from cattle trails with an average widths of 10-20 cm are present at the land use types pasture the slope steepness is steeper than it is considered by Wischmeier and Smith (1987) to use a suggested P factor. Thus, the variation of USLE results from calculations with $P = 1$ and $P = 0.9$ was considered within the range of error for all pastures. The cover and management factor C was separately determined for investigation areas with and without dwarf shrubs in dependence of the measured fractional vegetation cover (FVC) in the field (with FVC measurements of September). The following equations were used to implement the C factors to different fractional vegetation covers (US Department of Agriculture, 1977).
For investigation sites without dwarf shrubs we used:

$$C = 0.45 \times e^{-0.0456 x FVC} \qquad 3.7$$

for investigation sites with dwarf shrubs we used the following equation:

$$C = 0.45 \times e^{-0.0324 \times FVC} \qquad 3.8$$

The resulting C factors are listed in Table 1. The fractional vegetation cover was determined in April and September 2007 (Table 3.3). A grid of 1 m^2 was used with meshes of 0.1 m^2. The fractional vegetation cover of each mesh was estimated and averaged for the entire square meter. This procedure was repeated four times for each plot. The maximum standard deviation was ~ 5 %.

3.3 Results and Discussion

3.3.1 Cs-137 measurements and erosion rates

Mean Cs-137 activity was 91 Bq kg^{-1} (RSD ± 19.2 %) for hayfields, 94 Bq kg^{-1} (RSD ± 27.4 %) for pasture without dwarf shrubs and 121 Bq kg^{-1} (RSD ± 27.3 %) for pasture with dwarf shrubs (Table 3.2). The reference sites in the Urseren Valley had a mean Cs-137 activity of 146.4 Bq kg^{-1} (RSD ± 17.3 %). For our sites a mean soil bulk density of 1195 kg m^{-3} (RSD ±

13.7 %) was determined for the upper soil layer (0-10 cm) (Table 3.2). Average soil erosion rates of hayfields, pastures without dwarf shrubs and pastures with dwarf shrubs are listed in Table 2. The variability of erosion values on hayfields is quite high. The reason that could explain this variability might be high avalanche impact on hayfield hf3 (Ambuehl, 1961) and snow gliding processes on hayfield hf1 (Figure 3.2 right side). High avalanche frequency might cause higher erosion rates during winter time and early spring. In contrast, hayfield hf2 that has low erosion rates is neither influenced by avalanches nor by snow gliding. Hayfield hf2 and pasture with dwarf shrubs paw1 had similar erosion rates Even though the fractional vegetation cover was significantly lower in pastures (77 % paw1) than in hayfields (92 % hf2) (Tabel 3.3). The latter might be due to increased sedimentation by dwarf shrubs which stabilize the soil and act as physical barriers, thus, reducing transport of soil particles down slope.

Table 3.3: Fractional vegetation cover of all plots in April and September 2007 (RSD 5 %, *n = 3 times of vegetation cover determination of each plot).

	grassland type								
	hayfield (hf)			pasture without dwarf shrubs (pawo)			pasture with dwarf shrubs (paw)		
*vegetation cover [%]	hf1	hf2	hf3	pawo1	pawo2	pawo3	paw1	paw2	paw3
April 2007	77	93	76	65	62	67	77	79	73
September 2007	92	95	90	65	62	67	77	79	73

Mean annual erosion rates of pastures without dwarf shrubs are higher than erosion rates on pastures with dwarf shrubs (two to four times).

The measured values are higher than reported literature values of alpine grasslands (Felix and Johannes, 1995). The reported low measured rates of erosion in alpine regions have been determined during summer time. In contrast, measured erosion rates based on Cs-137 inventories in the Urseren Valley included extreme events and winter processes of 22 years. Thus, snow and avalanche activity is included and the likelihood to capture extreme events is much higher with our approach than with short term measurements and during summer time. Also, the high erosion rates up to > 37 t ha^{-1} a^{-1} are congruent with the high visible damages at the sites. Visible damages on alpine catchments can not be compared with visible damages on agricultural croplands, where large rills are visible after an erosion event took place. Large areas with low fractional vegetation cover and incoherent soil are visible at our sites. The latter are prone to erosion damages which are long lasting and continuous (Figure 3.2). Due to air photographs of the last 20 years it can be excluded that our high Cs-137 based erosion rates arose from land slides.

Figure 3.2: Eroded sites some days after snowmelt on hayfield hf1 (left hand side) and on pasture without dwarf shubs pawo2 (right hand side) in the Urseren Valley. A cattle trail can be seen in the upper right corner of the right picture. Both pictures illustrate visible erosion damages after winter time.

3.3.2 Erosion assessment with the USLE

Results of the USLE on the land use type pasture with dwarf shrubs are in a similar order of magnitude compared to Cs-137 based erosion rates if uncertainty of methods is considered (Figure 3.4). All pastures without dwarf shrubs and hayfields hf1 and hf3 are underestimated considerably by the USLE compared to Cs-137 based erosion rates. The reason for the underestimation of the latter land use types might be that winter processes like avalanches and snow gliding have a great influence on these plots. The damages on two of these plots just after snowmelt are shown in Figure 3.2a (pawo2) and 2b (hf1) whereas no damages could be seen on all paw plots (Figure 3.3). These winter processes are not considered within the USLE. The USLE erosion value for the land use type hf2 also fits to the Cs-137 based erosion values. Here again, no damages were visible after winter.
The uncertainty of the USLE is based on the measurement error of the plot steepness, due to the heterogeneity of grain size analyses as well as organic matter (Table 3.1) that was considered by the determination of the *K* factor with the nomograph of Wischmeier and Smith (1987), according to the measurement error of the fractional vegetation cover that was used for the determination of the *C* factor and to the uncertainty of the *P* factor. Generally, the *C* factors that were determined based on detailed information of the fractional vegetation cover on each single site lead to higher *C* factors compared to previous applications. The reason for the lower applied *C* factors by e.g. Prasuhn et al. (2007) was due to the general assumption that alpine grasslands have a fractional vegetation cover of minimum 95 %. Hence, *C* factors were estimated to be about 3×10^{-3} (chapter 3.1). Our USLE based erosion rates are, thus, also higher than previously reported (Table 3.1). So far, mean erosion values of the whole valley were assumed to be between 0 to 4 t ha^{-1} a^{-1}.

Figure 3.3: Investigation site pasture with dwarf shrubs (paw2) during spring time 2008. No damages of soil erosion are visible on the sites.

Our results indicate that the USLE is not suitable for steep alpine regions were snow processes have a great influence on total soil erosion amounts. Other erosion models like WEPP and PESERA have also been tested whether they are suitable to use in alpine regions (Konz et al., 2009; Meusburger et al., 2009). The WEPP model was found to work quite well for erosion simulation during summer time. However, erosion processes over longer time periods with winter processes were also underestimated by the model (Konz et al., 2009). General information of WEPP are given in Laflen et al. (1997) of PESERA in Kirkby et al. (2008)

Thus, the USLE was applied a second time for all nine plots in the Urseren Valley with a supplemental alpine factor *W*. This factor describes the failing information that led to the deviation of erosion results gained for the first run of the USLE. These deviations might be due to the influence of winter processes, measurement uncertainty or any other factors that are not yet known. All factors (*R K L S P C*) as well as the gained Cs-137 based erosion rates (*A*) were taken for the second run of the USLE. We inserted the W factor in the original USLE equation and solved to the alpine factor *W*. The gained W factors are listed in Table 3.4. This value can only be considered as first assumption of an adjusted USLE for alpine environments. More data is needed to validate such an alpine factor. However, this is a first step to solve the problem of erosion modeling in alpine environments. Due to the similarity of the W factor of the single land use types we might assume that this factor is mainly influenced by snow processes that have different effects on soil erosion amounts depending on the present land use types. These different influences on winter processes depending on the different land use types could be observed after snow melt on all land use types (Figures 3.2 and 3.3). Additionally, hayfield hf1 is visually most influenced by snow gliding, whereas hf3 is known to have a high frequency of avalanches during winter time (recorded by the Swiss Federal Institute for Snow and Avalanche Research Davos) lead to the highest alpine factors W.

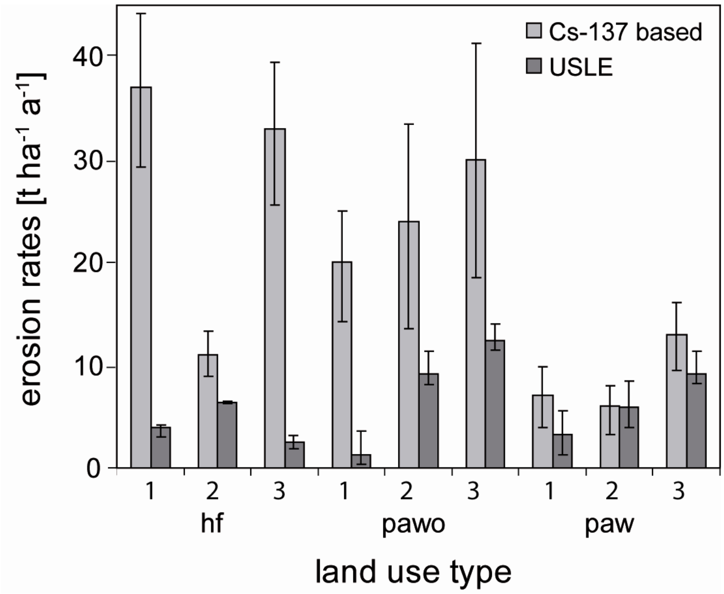

Figure 3.4: Erosion rates due to Cs-137 based measurements (light grey) and calculations with the Universal Soil Loss Equation (dark grey). Cs-137 error bars (17 %) are due to manually analysis of gamma spectra. USLE error bars are gained by consideration of the uncertainty of single parameters.

3.4 Conclusions and Outlook

We determined mean annual soil erosion rates of steep alpine slopes of 6-37 t ha^{-1} a^{-1} which are about 10 times higher than previously reported in literature. Our erosion rates were based on in-situ Cs-137 measurements and thus included winter processes as well as extreme rainfall events over the last 22 years since Cs-137 was released from the Chernobyl accident. These high rates of erosion are congruent with the high visible soil degradation in the Urseren Valley and explain the low recover rates of degraded alpine sites. Thus, we can conclude that soil erosion is a significant problem in alpine environments and has to be investigated in detail in the future for the development of soil conservation strategies. Erosion rates that are calculated with the Universal Soil Loss Equation are in the same order of magnitude compared to Cs-137 based erosion rates on the land use types pasture with dwarf shrubs (paw). On the land use types hayfields and pasture without dwarf shrubs, USLE based erosion values are considerably lower compared to the measured values. The high differences in erosion values are most likely attributed to the great visible damages in early spring just after snow melt. We suggest an alpine USLE including a alpine factor *W* that allows the user to implement a factor for slopes that are influenced by avalanches and snow gliding processes during winter time. An unambiguous statement can not yet be made due to our calculation since the data inventory is too low. Future research should aim to enlarge the data inventory also to other mountainous areas to provide a ground basis for an alpine USLE. This future work should include the investigation of known areas that are affected by either snow gliding processes during winter or by avalanches.

Acknowledgements

This work was financially supported by the State Secretariat for Education and Research (SER), in the framework of the European COST action no. 634: “On- and Off-site Environmental Impacts of Runoff and Erosion”. The authors thank H. Surbeck from the University of Neuchâtel, Switzerland for the supply of gamma-analysis software to analyse the NaI spectra and S. Estier from the Federal Office of Public Health (FOPH) for her valuable support and discussion.

CHAPTER 4

Process identification of soil erosion in steep mountain regions

Accepted by : Hydrology and Earth System Sciences
Vol.14, Issue 4, pp.675-686 , 2010
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4.1 Abstract

Mountainous soil erosion processes were investigated in the Urseren Valley (Central Switzerland) by means of measurements and simulations. The quantification of soil erosion was performed on hill slope scale (2 x 20 meters) for three different land use types: hayfields, pastures with dwarf shrubs and pastures without dwarf shrubs with three replicates each. Erosion rates during growing season were measured with sediment traps between June 2006 and November 2007. Long-term soil erosion rates were estimated based on Cs-137 redistribution. In addition, soil moisture and surface flow were recorded during the growing season in the field and compared to model output. We chose the WEPP model (Water Erosion Prediction Project) to simulate soil erosion during the growing season. Model parameters were determined in the field (slope, plant species, fractional vegetation cover, initial saturation level), by laboratory analyses (grain size, organic matter) and by literature study. The WEPP model simulates sheet erosion processes (interrill and splash erosion processes, please note that no rill erosion occurs at our sites). Model output resulted in considerable smaller values than the measured erosion rates with sediment traps for the same period. We attribute the differences to observed random gravity driven erosion of soil conglomerates. The Cs-137 measurements deliver substantially higher mean annual erosion rates, which are most likely connected to snow cover related processes such as snow gliding and avalanche activities.

4.2 Introduction

Soil erosion is a major environmental problem in many parts of the world (Nearing et al., 2004). The dominant processes in agricultural lowlands such as splash, rill and interrill erosion are well investigated in numerous studies over the last decades (e.g. Govers et al., 1999; Hessel et al., 2003; Walling and He, 1999). However, less attention has been paid to erosion processes in mountainous systems, where the extreme climate and steep slope angles trigger soil erosion processes. Gravity forcing is a crucial process in these environments in combination with animal activity during the growing season. Rill erosion processes are not observed in unploughed mountainous environments. In the following the term sheet erosion summarizes erosion through unconcentrated flow (often referred to as interrill erosion) and splash erosion processes. In addition, snow cover related mechanical friction and/or abrasion processes will occur during winter time and may also have a significant influence on erosion rates in mountainous environments. Water induced soil erosion in mountainous regions is greatly influenced by land use and management as well as by climate, extreme topography and soil erodibility (Alewell et al., 2008; Simonato et al., 2002). Following the above discussion, the term soil erosion is not referring to landslides or rapid mass movements in shallow soils in this study.

Since snow dynamics make it difficult to measure soil erosion in mountainous regions throughout the year, most of the erosion measurements have been conducted during the growing season without the influence of snow (Descroix and Mathys, 2003; Felix and Johannes, 1995; Isselin-Nondedeu and Bedecarrats, 2007). Erosion rates in alpine grasslands measured by Felix and Johannes (1995) were between 0.1 and 10 kg ha^{-1} for the growing season. Isselin- Nondedeu and Bedecarrats (2007) measured the influence of several plants on soil erosion. They found considerable difference with Festuca Alpina having the highest amount of sediment deposition.

Climate change has an effect on the increase of thawing days in mountainous regions (Appenzeller et al., 2008). Snowmelt is reported to occur earlier in spring due to rising temperatures (Laternser and Schneebeli, 2003). This indicates a higher amount of precipitation in the form of rain compared to snow. Surface runoff in winter and spring is predicted to be higher with potentially increasing soil erosion during times of sparse or no vegetation cover (Fuhrer et al., 2006). A better understanding of mountainous erosion processes is therefore a prerequisite for all types of land use and climate change assessment studies and the development of mitigation concepts.

This study aims at determining the dominant erosion processes in mountainous environments. The assessment is based on a combined observation and modeling approach. We used the Water Erosion Prediction Project Model (WEPP) because it is a well established tool to simulate water erosion and sediment yield. WEPP has been applied in various geographic locations across the United States (e.g. Huang et al., 1996; Laflen et al., 2004; Savabi et al., 1993), in Australia (Yu and Rosewell, 2001) and in Europe (Brazier et al., 2000; Gronsten and Lundekvam, 2006; Pieri et al., 2007; Raclot and Albergel, 2006). The application of WEPP in steep mountainous environments has been tested once in the Italian Alps by Simonato et al. (2002). The authors successfully reproduced measured erosion rates. Measurements of erosion rates were done with sediment traps during the growing season and mean annual erosion was measured by means of Cs-137 activity over the last 22 years to obtain additional information of erosion processes for mountainous regions. Our hypothesis was that the differences between the model results and the measurements can be attributed to processes specific for mountainous regions that are less relevant in low lands and thus not covered by the model.

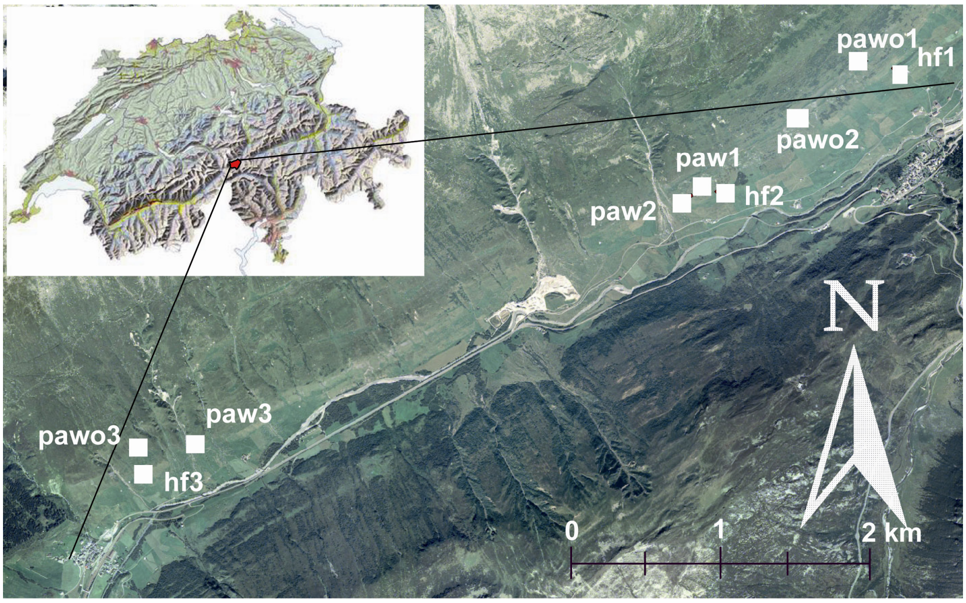

Figure 4.1: The Urseren Valley in Central Switzerland and location of the investigation sites with three land use types: hayfield (hf), pasture without dwarf shrubs (pawo) and pasture with dwarf shrubs (paw). The meteorological station is on the valley bottom at an elevation of about 1400 meter a.s.l., 1 km north-east of measurement station hf1.

4.3 Materials and Methods

4.3.1 Investigation area

The study area is located in Central Switzerland (Canton Uri) in the Urseren Valley (Figure 4.1). The sub-catchment of the Furka Reuss has an area of about 30 km^2. The elevation of the WE oriented mountain valley ranges from about 1400 m a.s.l. to about 2500 m a.s.l. Measured mean annual rainfall from 1986 to 2008 is 1516 mm, and mean air temperature at the weather station (1400 m a.s.l.) is 3.1°C (MeteoSwiss, 2007). The valley is snow-covered from about November to April with the maximum snow height occurring usually in March. The mean annual snowfall from 1986 to 2008 is 448 mm water equivalent. Discharge is usually dominated by snowmelt from May to June. Important contributions to the flow regime are early autumn floods.

The dominant land use types in the valley are hayfield with hay harvesting near the valley bottom, and pasture further upslope. Siliceous material is dominant, forming cambic podzols (anthric) (IUSS Working Group, (2006). Vegetation shows strong anthropogenic influences due to centuries of pasturing (Kaegi, 1973). The characteristic of these soils is a migration (M-horizon) horizon within the upper about 100 cm that is caused by sedimentation in the past. The thickness of the M- horizon is between 5 and 45 cm.

4.3.2 Experimental plots

The experimental plots (~40 m^2) are situated at the south-facing slope at an altitude of 1550 to 1800 m a.s.l. (Figure 4.1). Three different land use types with three replicates each were investigated. The land use types are hayfield (hf1, hf2, hf3), pasture with dwarf shrubs (paw1, paw2, paw3) and pasture without dwarf shrubs (pawo1, pawo2, pawo3). The slopes of all plots were in the range of 35° to 39°. Soil textures of all 9 plots are listed in Table 4.1. The hayfield vegetation is dominated by Trifolium pratense ssp. Partense, Festuca sp., Thymus serpyllum and Agrostis capillaries. Pasture with dwarf shrubs are dominated by Calluna vullgaris, Vaccinium myrtillus, Festuca violacea, Agrostis capillaries and Thymus serpyllum. Dominant plant species on pastures without dwarf shrubs are Glubelaria cordifolia, Festuca sp. and Thymus serpyllum.

Quantification of soil erosion

Soil erosion rates were measured with two independent techniques. Sediment traps enabled the direct measurement of erosion rates during the growing season. Long-term erosion rates can be estimated by Cs-137 isotope analysis as indirect method. This method considers the redistribution of Cs-137 after its deposition in the year 1986 by the Chernobyl accident. The Cs-137 method delivers an integral value including winter and growing season processes.
Sediment traps
Sediment traps for erosion rate measurements were installed at each plot in July 2006 (Figure 4.2) to measure erosion rates during the growing season. A geotextile which is fixed to the ground was used to detain the transported particles (Robichaud and Brown, 2002). The sediment trap was equipped with a v-shaped steel plane below the geotextile to collect and quantify the surface water flow (Figure 4.2c). The quantification of the overland flow was done with a two-bowl tipping bucket (Figure 4.2d). Each bowl had a capacity of 0.5 liter (EnvironmentalProducts, 2006).
Material that is transported into the geotextile was collected and weighted every second or third week during the growing season from April to November. In addition, at one plot of each land use type (hf3, pawo2, paw2), precipitation and soil moisture were measured continuously every 10 minutes. Precipitation was measured with tipping buckets (ECRN-50 rain gauge, DecagonDevices), soil moisture was measured with an EC-5 sensor (precision ± 2%), (DecagonDevices). All data were logged by means of an Em50 Data Logger (DecagonDevices).

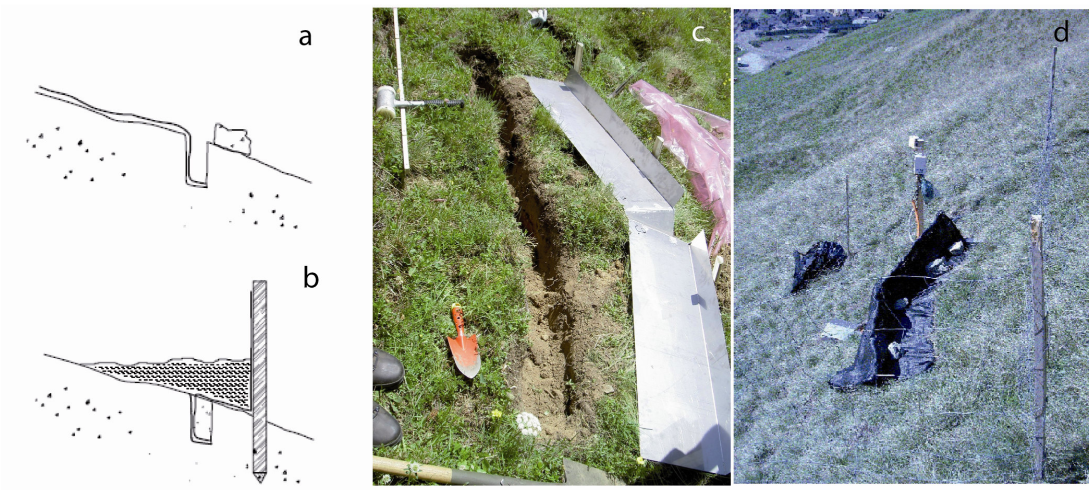

Figure 4.2 : (a+b) Sediment trap after Robichaud et al. (2002) and (c) extended version under construction in the Urseren Valley (2006) with a steel plate to concentrate the surface flow. The steel plate was finally attached to the upper boundary of the filled trench where the geotextile comes out of the trench. (d): Completed sediment trap for erosion measurement at land use type hayfield (hf3).

It is likely that the used sediment traps tends to underestimate the surface water flow. The underestimation is due to small gaps between soil and geotextile at the upslope edge of the sediment trap inlet. Surface flow might therefore partially trickle away. However, we installed the sediment traps in July 2006 one year before the beginning of the experiments. This ensured the recovery of the soil edges and the surface cover and, therefore, mitigated the above mentioned error. The measurement period presented in this study of the sediment traps was from 2nd of April 2007 to 1st November 2007.

Cesium-137

As sediment traps provide information on total erosion for single growing seasons, long-term information of soil erosion of all plots was obtained from Cs-137 measurements conducted in autumn 2007. Cs-137 measurements for the determination of soil erosion rates since the fallout of Cs-137 (Chernobyl accident in April 1986) is a common in the low lands (e.g. Ritchie and McHenry, 1990; Walling and He, 1999), but has only been recently used in subalpine terrain (Konz et al., 2009). We used a NaI scintillation detector for Cs-137 measurement on our nine subalpine test plots. For the measurement procedure, the NaI scintillation detector was placed perpendicular to the ground at a height of 25 cm and measured for 1h. To estimate the erosion rate from Cs-137 measurement we require the depth distribution of the Cs-137 concentration within the soil. The depth distribution was measured for a soil core taken at the reference sites. The soil core was portioned into slices of 2.5 cm thickness and the Cs-137 activity was measured in the laboratory with a GeLi-detector. We found that the Cs-137 concentration decreases logarithmically with depth, whereby the concentration reduces by the half about every 5 cm. The reference sites were chosen at places that are influenced neither by erosion nor deposition. Both reference sites are located at the valley bottom. Deposition of eroded soil on the reference sites can be excluded because of lateral moraines between the steep slopes and the reference sites. Soil erosion on both reference sites is unlikely due to a constant 100 % vegetation cover since 1986. This can be confirmed from air photographs that were taken regularly since 1986. Additionally, the slope of both reference sites is 0 %. For further details please see Konz et al. (2009).

4.3.3 Description of WEPP

WEPP is a physically based simulation model (Flanagan and Nearing, 1995) that describes mechanisms controlling water induced erosion including anthropogenic impacts such as irrigation, grazing, cutting and ploughing. The hill slope version (v2008.907) of WEPP contains nine components: a weather generator, snow accumulation and ablation, irrigation, surface and subsurface hydrology, plant growth, residue decomposition, soils and erosion. The determined vegetation as well as variable stocking rates and the cattle trails (variable configuration of slope intersections) can be transcribed by the model. Furthermore, the winter hydrology component is designed to simulate snow accumulation, snow density, snowmelt, and soil frost and thaw, all on an hourly basis (Savabi et al., 1995). The WEPP model is well tested in low land applications and is suitable to simulate dominate erosion processes of the low lands. Mountain specific processes, e.g. animal induced gravity forcing and snow mechanical processes cannot be handled by the WEPP model. Thus, a comparison of simulation results with measured erosion rates by Cs-137 can not be expected due to the missing processes. However, the simulations can be used to separate the bulk erosion rates into sheet erosion (splash and interrill erosion) and mountain specific processes.

WEPP inputs

Four modules of the WEPP model can be modified by the user (delivering input information for the nine components that are described above). These four modules are climate (rainfall amount, duration and intensity of rainfall, wind velocity and direction, temperature, solar radiation and dew point temperature), slope, soil (albedo, initial water saturation, interrill and rill erodibility, critical shear parameter, hydraulic conductivity, cation exchange capacity and organic matter (Table 4.1) and management. For the climate description, field-observed precipitation, daily temperature, solar radiation and wind (velocity and direction) were used. The data were taken from the meteorological station located at the valley bottom (1400 m a.s.l.), whereas the investigation areas are at the south-facing slope (at about 1650 m a.s.l). Hence, temperature at single south facing plots is slightly higher (up to 2°C, depending on the sky cover) than at the valley bottom due to the increased incoming short-wave radiation on the inclined surfaces.
The soil properties soil texture, cation exchange capacity (CEC) and organic matter content were determined for the first 50 cm (0-10 and 10-50, Table 1) by laboratory measurements. Critical shear stress (τc), interrill erodibility (Ki), rill erodibility (Kr) and hydraulic conductivity were calculated based on equations of the WEPP User Summary (Flanagan and Livingston, 1995) depending on grain size analyses that were measured with 10 replicates at each plot (Table 4.1). An initial water saturation degree was set for all plots at 25% in January 2007, based on soil moisture measurements. As there is no rotation of management type and plant composition in this investigation area, one management type was assigned for each land use type, as well as one composition of plants for the entire period. The surface of grassland does not have the typical rill and interrill pattern that leads to the defined rill and interrill erosion. This process has been realized by adjusting the random roughness (range management file) based on field measurements. Thus, WEPP simulations are concentrated on interrill erosion (sheet erosion). For the initialisation of the model's storages and bio-activity simulations an artificial warming up period was constructed using 20 times the data of the year 2007. The model output stabilized after around 5 to 6 years (Figure 4.3). The slight fluctuations in erosion rates are due to non-annual biomass cycles simulated by the model.

Table 4 1a: Soil parameter for the three investigated meadows (grain size analyses are given in % weight of fine-grained soil < 2000µm). The maximum standard deviation is 10% for grain size analyses, 9.5% for organic matter, 4.8% for pH value, 5% for fractional vegetation cover and 4.5% for slope steepness (for all three Tables 4-1a, b and c).

	m1		m2		m3	
depth [m]	0-0.1	0.1-0.5	0-0.1	0.1-0.5	0-0.1	0.1-0.5
sand (63-2000 µm) [%]	31.9	33.1	23.8	22.1	37.2	34.1
silt (2-63µm) [%]	42.3	39.8	45.5	43.9	41.4	33.6
clay (< 2µm) [%]	13.7	16.1	15.3	13.8	16	12.6
organic matter [%]	12.9	6.7	12.2	6.1	12.8	6.4
pH value	5.0	na	4.4	na	4.5	na
fractional veg. cover [%]	92		95		90	
slope [°]	39		36		39	

Table 4.1b: Soil parameter for the three investigated pastures without dwarf shrubs.

	pawo1		pawo2		pawo3	
depth [m]	0-0.1	0.1-0.5	0-0.1	0.1-0.5	0-0.1	0.1-0.5
sand (63-2000 µm) [%]	24.6	24.3	25.2	22.1	27.4	28.2
silt (2-63µm) [%]	38.2	37.8	32.4	34.6	30.1	31.4
clay (< 2µm) [%]	12.1	14.6	11.3	11.8	10.9	11.2
organic matter %	12.6	6.3	12.8	6.4	12.2	6.1
pH value of soil	7.1	na	7.3	na	4.6	na
fractional veg. cover [%]	65		62		67	
slope [°]	38		38		35	

Table 4.1c: Soil parameter for the three investigated pastures with dwarf shrubs (paw).

	paw1		paw2		paw3	
depth [m]	0-0.1	0.1-0.5	0-0.1	0.1-0.5	0-0.1	0.1-0.5
sand (63-2000 µm) [%]	23.1	27.0	22.6	25.9	28.6	31.5
silt (2-63µm) [%]	52.3	49.8	55.7	47.9	49.3	52.2
clay (< 2µm) [%]	8.7	8.5	9.5	7.6	11.2	10.1
organic matter [%]	11.9	6.1	11.9	6.0	12.2	6.1
pH value of soil	4.3	na	4.4	na	4.5	na
fractional veg. cover [%]	77		73		79	
slope [°]	38		38		35	

Application of the WEPP model

The calibration, we applied an experience based approach (Konz et al., 2007), takes our process knowledge gained from the measurement plots into account. Thus, an initial model parameter set was estimated according to measured system characteristics (e.g. soil texture, climate parameters, relief, fractional vegetation cover), available data from literature (e.g. rooting depth) and the parameter sets derived from earlier WEPP-applications in other basins (for the plant specific parameters). However, due to heterogeneity of soils and our inability to measure all model scale parameters (e.g. effective hydraulic conductivity that is derived from grain size analyses) the physically measured soil parameters usually cannot be taken one-by-one for the simulation. We therefore conducted a sensitivity analysis by varying one parameter and keeping the others fixed in order to identify the most sensitive parameters. The most sensitive parameters (precipitation amount and intensity, grain size, temperature, slope, canopy cover, random roughness, effective hydraulic conductivity) were considered in a Monte-Carlo analysis by allowing them to vary within reasonable ranges, which were defined by measurement uncertainty. 10,000 runs were computed and the simulation results of overland flow and erosion were analyzed relative to the initial model parameter set. We found that the initial model parameter set, which is based on measurements and literature values, delivered reliable simulation results within the uncertainty ranges given by the Monte-Carlo simulations and we therefore used this parameter set for the simulation of the erosion rates. In fact, the model was most sensitive to changes in canopy cover and precipitation intensity allowing them to vary in a broader range. However, within the parameter range constrained by estimated measurement errors the sensitivity of the parameters significantly declined and with it the parameter uncertainty.

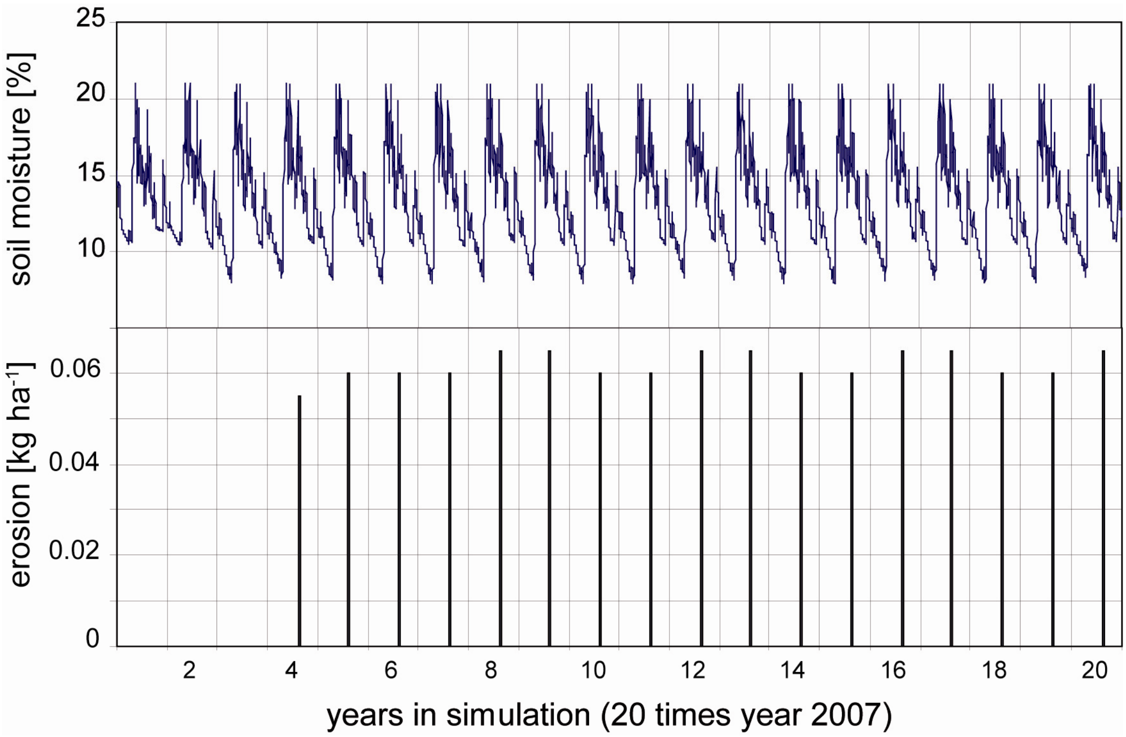

Figure 4 3: Initialization of the WEPP model for the land use type hayfield for 20 times 2007. The same year was taken 20 times instead of a time series in order to identify the stabilization of the water amount and erosion processes independent of the variability of water amount from precipitation.

This analysis was conducted to assess the predictive power of our initial model parameter set and it should be mentioned that it is not considered to be a thoroughly parameter uncertainty analysis. Please note, the model application is seen as a screening exercise to delineate sheet. WEPP does not simulate mountain specific processes. Therefore, we considered the experience based calibration, supported by Monte-Carlo analysis, as a suitable tool to differentiate between sheet erosion rates (simulated with WEPP) to total rates during growing season (sediment trap measurements) to total whole year erosion rates (Cs-137 measurements).

4.4 Measurement Results

4.4.1 Measured erosion rates during the growing season

Erosion rates measured with sediment traps yielded in comparable results during 2006 and 2007. Two dominant erosion processes were identified during the growing season: (i) relocation of grains by sheet erosion processes and (ii) soil conglomerate movement triggered by animal trampling and followed by gravity forcing. Total monthly erosion rates ranged from 0 to 4.4 kg ha^{-1} for hayfields (hf), from 1 to 68 kg ha^{-1} for pasture without dwarf shrubs (pawo) and from 1 to 11 kg ha^{-1} for pasture with dwarf shrubs (paw) (Table 4.2).

Table 4 2: Monthly measured (meas.) and WEPP-simulated (simul.) erosion rates (kg ha^{-1}) for the growing season April to November 2007 for the investigated three land use classes hayfield (hf), pasture without dwarf shrubs (pawo) and pasture with dwarf shrubs (paw). About 90 % of all measured erosion rates is caused by gravity forcing. Erosion values that are due to sheet erosion (overland flow and splash erosion) are given in brackets behind the erosion values. For pawo it is even more than 95%.

	land use type					
	hf3		pawo2		paw2	
	meas.	simul.	meas.	simul.	meas.	simul.
April	0 (0)	0	39 (<1.9)	0	1 (0.1)	0
May	4.4 (0.4)	0.2	44 (<2.2)	0.3	8 (0.8)	0.2
June	1.3 (0.1)	0.1	22 (<1.1)	0.2	5 (0.5)	0.1
July	0.5 (0.05)	0	68 (<3.4)	0	11 (1.1)	0
August	1 (0.1)	0	62 (<3.1)	0.3	3 (0.3)	0
September	0 (0)	0	2 (<0.1)	0.1	3 (0.3)	0
October	0 (0)	0	1 (<0.05)	0	1 (0.1)	0

Soil conglomerates were observed regularly in the sediment traps with diameters up to 30 cm (Figure 4.4) during field observations in 2007 mostly on the land use type pasture without dwarf shrubs. These eroded soil pieces cannot be explained with the movement of soil particles through overland flow and splash erosion but rather by animal activity and the steepness of the slopes, where soil conglomerates are subject to gravity forcing. In order to separate between the two erosion processes we weighted the conglomerates and the fine soil material separately. This separation process resulted in a fraction of soil from sheet erosion of about 5-10% (Table 4.2) compared to a much larger fraction of 90-95% due to gravity forcing process. The method tends to even overestimate sheet erosion rates because soil particles detach from the conglomerations while falling into the trap. The highest difference of total erosion rates can be observed at the land use site pasture without dwarf shrubs. Reasons for this could be the use of rangeland with higher trampling damage. Dwarf shrubs obviously reduce sediment transport. This can be explained with the hindering effect of dwarf shrubs on the transport of soil particles (Konz et al., 2009). Hayfields seem to be generally less susceptible to sheet erosion than pastures during the growing season. In general, erosion rates are very low, compared to observations in agricultural low lands.

Figure 4.4: Soil erosion during the growing season 2007. Large soil conglomerates were collected in the sediment traps ranging from 1 to 30cm.

4.4.2 Long term erosion rates

The long term Cs-137 erosion rates expressed as mean annual rates were about three magnitudes larger than the rates observed for the growing season at all nine plots (Figure 4.5). It is very unlikely that inter annual variability and single high erosion years/events on specific plots are responsible for the observed discrepancy between the long term averaged high annual erosion rates and the measured low erosion rates during to growing seasons. Instead, it is very likely that winter processes such as freezing and thawing and snow cover dynamics trigger high erosion rates.

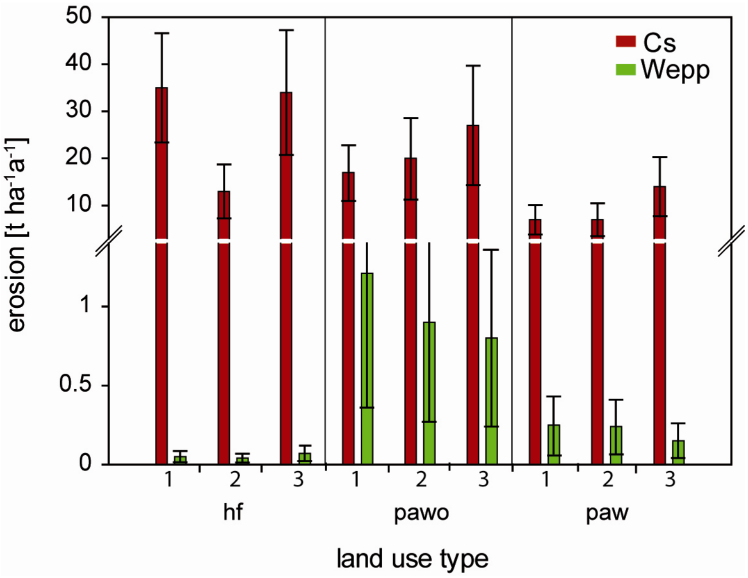

Figure 4.5: Mean annual erosion rates of WEPP simulation compared to Cs-137-based erosion rates of all investigation plots for the period 1986-2007. Cs-137 error bars are due to manual analysis of gamma spectra (17%), the heterogeneity of each single plot (n=3; mean standard deviation 10.1%) and uncertainty of soil porosity. WEPP errors result from sensitivity analysis.

Contradictory to the seasonal erosion rates the highest annual rates (~35 t ha^{-1}a^{-1}) could be found on hayfields. Mean annual erosion rates of pastures without dwarf shrubs are higher than erosion rates on pastures with dwarf shrubs (two to four times). This observation is congruent with the growing season measurements.

The variability of the long term mean annual erosion values on hayfields (hf1-hf3) is quite high ranging from 10 t ha^{-1} a^{-1} up to 37 t ha^{-1} a^{-1}. In contrast, the variability between the three replicates of pasture with dwarf shrubs (paw1-paw3) and pasture without dwarf shrubs (pawo1-pawo3) is lower (ranging from 7 t ha^{-1} a^{-1} up to 13 t ha^{-1} a^{-1} for paw and from 20 t ha^{-1} a^{-1} up to 29 t ha^{-1} a^{-1} for pawo).

The reason for the high spatial and temporal variability of the hayfields´ erosion could be the high avalanche activity on the hayfield sites (Ambuehl, 1961). Furthermore, we observed a high influence of snow gliding processes (see Figure 4.6). Ambuehl (1961) determined the avalanche activities on the slopes of the Urseren Valley and developed an avalanche map that was compared with our study sites. Additional regular field visits and qualitative avalanche monitoring (simple counting of avalanches on the specific sites) during winter 2007/2008 confirmed Ambuehl´s results. The site hf3 experiences regularly a high avalanche activity. Neither avalanches nor snow gliding processes were observed on hf2 by Ambuehl (1961) and during our investigation period.

Figure 4.6: Investigation site pawo2 (left hand side) and hf1 (right hand side) shortly after snow melt. This picture gives an impression of the possible influence of snow and 'snow gliding' on erosion processes.

4.5 Simulation results of the WEPP model

As discussed in Section 4.3.6 the WEPP model simulates the sheet erosion, namely splash erosion and interrill erosion (rill erosion was not modeled because of the failing rill-interrill shape). In order to assess the model performance and the reliability of the simulated erosion amounts we can compare results of the hydrology modules (overland flow and soil moisture) to plots measurements (hf3, pawo2, paw2).

4.5.1 Hydrology

The hydrology (overland flow and soil moisture content) was simulated well during the growing season from April to October 2007 (Figure 4.7). Simulated overland flow compares well with the measurements for all land use types (for example see simulation results and measurements of the land use type hayfield, Figure 4.7). Overestimations of overland flow occurred in May, June and August (42%, 33% and 53%, respectively). The sediment traps with equipment for overland flow measurement tend to underestimate the surface flow (see section 4.5). Thus, the simulation bias falls within the expected error of the observed flow rates. Interestingly, in July, September and October surface flow is observed but not simulated.

The dynamic of the observed soil moisture is reproduced well for all land use types (Figure 4.7) from the end of April onwards and even very specific patterns of the soil moisture dynamics are simulated well for the land use type pasture without dwarf shrubs (pawo).

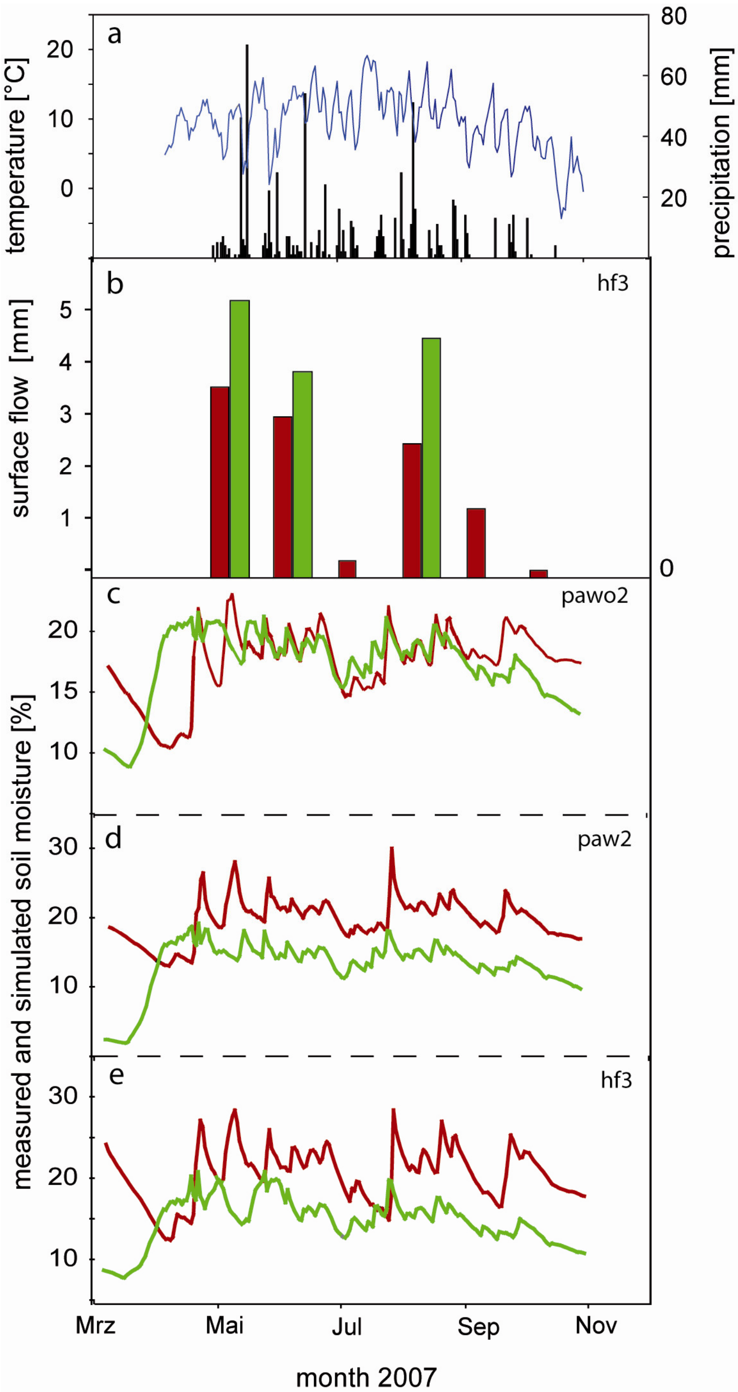

Figure 4.7: (a) measured daily precipitation (black) and mean air temperature (blue), (b) measured (red) and simulated (green) surface flow for hf3. (c-e) measured (red) and WEPP simulated (green) soil water content for April to November 2007 for one of all three land use types pasture without dwarf shrubs (pawo2), pasture with dwarf shrubs (paw2) and hayfield (hf3) for the first 35cm.

A general underestimation of about 5% during summer time can be observed for the land use types pasture with dwarf shrubs (paw) and hayfields (hf), but dynamics are reproduced satisfactory throughout the entire growing season. A possible reason could be an overestimation of evapotranspiration with values of about 2.4 mm day^{-1} (mean values during growing season) for hayfields and 3.2 mm day^{-1} for dwarf shrubs (mean values during growing season). Fecht et al. (2005) provides values of about 1.7 mm day^{-1} (mean values during growing season) for hayfields and 2.8 mm day^{-1} (mean values during growing season) for dwarf shrubs in mountainous environments.

Significant discrepancies between measurements and observations of soil moisture both in terms of dynamics and absolute values can be observed in the end of March and beginning of April (Figure 4.7c-e). Measurements and simulations show contradictory patterns with increasing simulated soil moisture and decreasing measured soil moisture. Reasons for that could be wrong assumption of snow accumulation and ablation calculations in WEPP (Figure not shown). However, the model performance was satisfying during the major part of the growing season when snow accumulation and ablation processes did not have any influence (Figure 4.7c-e).

Since the hydrology dominates the erosion processes a reliable simulation of overland flow is a prerequisite for erosion predictions. The WEPP model reproduces the overland flow rates with acceptable accuracy and therefore provides reliable inputs for the erosion module. Although the model overestimates the observed runoff rates it should be considered that the measurement of overland flow is subject to measurement errors (underestimation, see chapter 4.3.2). Thus, the simulation results are considered to be within a range that is satisfying and can be considered as appropriate for erosion simulations. Since the erosion processes are simulated with equations which only use surface water level as transient variable the hydrological input is the most important input that can be related to internal model performance assessments. The additional information required for erosion simulations, e.g. land use and soil parameters, are externally derived parameters and the errors caused by those values cannot be considered as internal model uncertainties. We therefore consider the WEPP model simulations as reliable and suitable to reproduce the erosion rates caused by the sheet erosion processes during the growing season.

4.5.2 Simulated soil erosion rates during the growing season

Table 4.2 compares the simulated and the measured sheet soil erosion rates during the growing season 2007. The measured values of sheet erosion rates (without conglomerates due to gravity forcing) were generally very low (Table 4.2). The model simulates erosion rates in the same order of magnitude for all three investigated land use types. As given in Table 4.3 only the growing season of paw2 shows significant differences compared to our observation year 2007. The 2007 simulation delivers a sheet erosion rate of 0.3 kg ha^{-1}a^{-1} for the growing season compared to the simulated mean long term value of sheet erosion of 6.4 kg ha^{-1}a^{-1} (value of Table 4.3, 140 kg ha^{-1} divided by 22 years). This deviation is most likely due to extreme precipitation events that have a higher influence on erosion processes on pastures due to less vegetation cover. No extreme rainfall events occur during our measurement period (Figure 4.8). However, even if we consider the simulated mean long-term erosion value for paw2 it is significantly lower than the measured total erosion rate during growing season in 2007 (32 kg ha^{-1}, sum of paw2 in Table 4.2). Therefore, the model proves the right magnitude of the measured sheet erosion rates for all land use types. The simulated mean annual rates are significantly higher than the corresponding rates of the growing season.

Generally, our measured soil erosion rates during the growing season are low but comparable to the measured erosion rates of Felix and Johannes (e.g. Felix and Johannes, 1995) (0.1 to 200 kg ha^{-1}) in the Berchtesgaden region (South Germany, Bavarian Alps). They conclude that their low erosion rates are based on low effective precipitation that is between 1 and 2%. The effective precipitation on our investigation plots is comparable and ranges between 0.6% and 2% during the growing season. Frankenberger (1995) measured much higher erosion rates during the growing season up to 20000 kg ha-1. However, he reported that the effective precipitation during the investigation period was up to 60%.

Table 4 3: Comparison of simulated erosion rates 1986-2007 during the vegetation periods (April-October) and simulated erosion rates including the whole year (January-December) from 1986-2007. Erosion rates are given in t ha^{-1}.

	land use type					
	hf3		pawo2		paw2	
	growing season	complete period	growing season	complete period	growing season	complete period
Σ 1986 -2007	5	154	40	20900	140	4400

Extreme erosion rates (low as well as high rates) are generally challenging for soil erosion modeling (Nearing, 1998). Thus, the inaccuracy of simulated erosion values is a general problem of models and not a specific problem of the WEPP model. This is shown in the study of Simonato et al. (2002). They did erosion simulations based on the WEPP model and the RUSLE that were compared to plot measurements in the Italian Alps. The study was done during the growing season 1998 and 1999. The lowest measured values were 0 kg ha-1, highest values 3000 kg ha-1. For those plots WEPP simulations resulted in erosion rates of 210 kg ha-1 and 4720 kg ha-1, respectively. Results of the RUSLE are 39000 kg ha-1 and 41000 kg ha-1, respectively. Based on this study the WEPP model seems to be more useful for the prediction of low mountainous erosion rates during the growing season. Moreover, it could be shown that some further studies confirm low erosion rates during the growing season in mountainous regions (Felix and Johannes (1995); Simonato et al. (2002)).

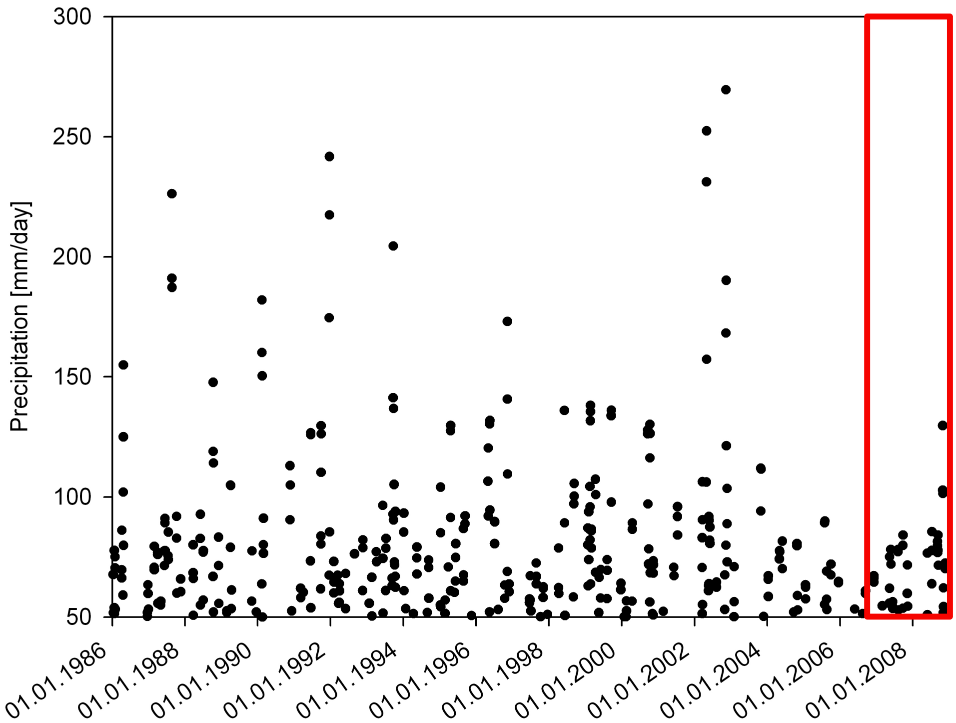

Figure 4 8: Daily sums of precipitation (>50mm) from 1986 to 2008. Red box marks the time period were erosion rates were measured with sediment traps.

4.6 Relative contribution of soil erosion processes

Although a direct comparison of measured long term and seasonal data is difficult due to the different time windows of observation (the long term annual value is the average erosion rate of 22 years compared to the seasonal data of seven months), this comparison gives a good indication of the dominant erosion processes at our subalpine sites. From the long-term measurements, the sediment trap measurements, and the modeling exercises, we can distinguish between three major process classes: (i) the direct measurements of gravity forcing processes and sheet erosion processes during growing season, (ii) the modeling of sheet erosion and (iii) the indirectly inferred contribution of winter processes delineated from the difference between Cs-137 derived erosion rates and modeled erosion rates (Figure 4.9). Based on our combined measurement and simulation analyses we can provide first quantitative measures of the relative contributions of each sub-process (Table 4). Erosion processes during the growing season are negligible small compared to the winter processes. Gravity forcing and development of soil conglomerates due to animal activity dominate the growing season. Since the highest erosion rates were measured on slopes that are reported to have a high avalanche risk or are prone to snow gliding during winter time (hf1, pawo3 and hf3) it is very likely that those processes are responsible for the high erosion rates. Furthermore, the discrepancy between the high Cs-137 based erosion rates and the measured or simulated rates are most likely attributed to winter processes because (i) we measured summer processes and (ii) these winter processes are not implemented in WEPP, thus explaining the failure of the model. However, the influence of snow mechanical processes is so far poorly understood and has to be investigated in more detail in the future.

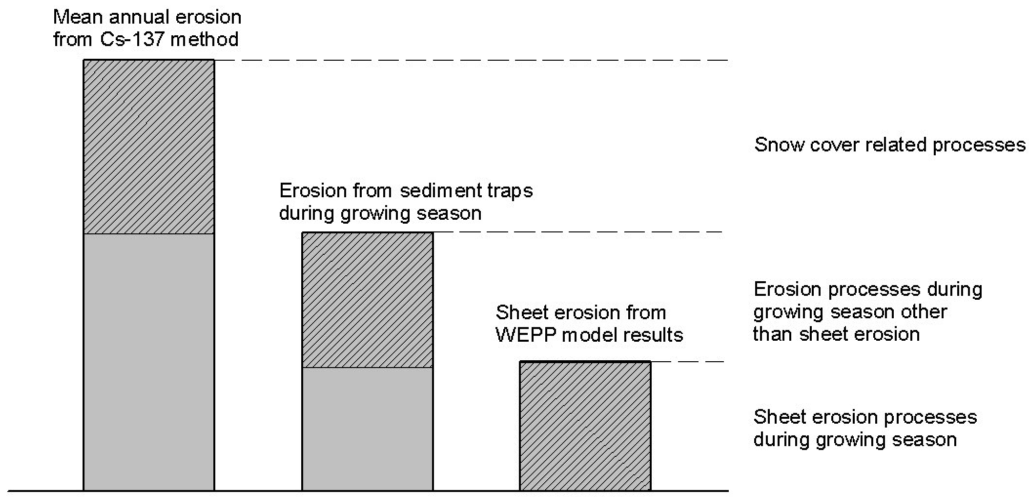

Figure 4 9: Conceptual results of process identification approach; marked parts of the bars indicate the respective processes, whereas the entire bar shows the total measured value. The process separation was done by calculating the relative differences between the annual rate, the growing season measurements and the model simulations during the growing season.

Table 4 4: Relative contributions of the three process classes identified for the Urseren Valley.

	hf3 (%)	pawo2 (%)	paw2 (%)
Gravity forcing	0,019	1,13	0,36
Sheet erosion	0,0019	0,06	0,04
Winter erosion	99,98	98,81	99,60

4.7 Conclusions

We distinguished between the dominant erosion processes in mountainous environments. The low measured erosion rates were confirmed by simulations with the physically based model WEPP. We consider the WEPP simulations to be reliable because the driving transient hydrological variables are reproduced with acceptable accuracy. WEPP failed to capture the high mean annual (1986-2007) erosion rates as indicated by Cs-137 measurements. We interpreted the latter differences to be due to winter processes which are not included in the WEPP simulation. Besides sheet erosion processes, random gravity driven movements of conglomerates following animal trampling could be distinguished. While gravity driven soil erosion rates were already 20 times higher than sheet erosion rates, but the by far highest fraction (~99%) of erosion is caused by winter processes as identified by Cs-137 measurements. The winter processes were dominant for all our nine investigation plots. Therefore it is unlikely that single high erosion events are responsible for the high values.
Regarding model performance we conclude that the model reproduces sheet erosion processes and can therefore be used to differentiate between confounding factors of erosion in mountain systems but cannot be used to simulated whole year erosion rates in systems where winter processes dominate erosion rates. This WEPP application was the first comparison between high temporal resolute field installations (erosion, soil moisture, and surface flow measurements) and WEPP simulations in mountainous areas.
The analysis of measurements and simulations indicated that winter processes have to be investigated further; most likely avalanche activity and snow gliding processes are mainly relevant for sediment detachment and transport. Future research should concentrate on interface processes between soil and snow cover, e.g. friction and abrasion. As stated the numbers presented here are first estimates and need further measurements to prove the generality of our results.

Acknowledgements

This work was financially supported by the State Secretariat for Education and Research (SER), in the framework of the European COST action no. 634: “On- and Off-site Environmental Impacts of Runoff and Erosion”.

CHAPTER 5

Application of in-situ measurement to determine 137Cs in the Swiss Alps

Accepted by: Journal of environmental radioactivity
Vol. 101, Issue 5, pp. 369-376, 2010
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5.1 Abstract

In-situ measurements in steep alpine environments have not often been done. Most studies have been carried out in arable lands and with Ge detectors. However, NaI detector system is an inexpensive and easy to handle field instrument. In this study, a comparison of laboratory measurements with GeLi detector and in-situ measurements with NaI detector of ^{137}Cs gamma soil radiation has been done in an alpine catchment (Urseren Valley, Switzerland). The aim of this study was to calibrate the in-situ NaI detector system for application on steep alpine slopes. Replicate samples from an altitudinal transect through the Urseren Valley, measured in the laboratory with a GeLi detector, showed a large variability in ^{137}Cs activities at a meter-scale. This small scale heterogeneity determined with the GeLi detector is smoothed out by uncollimated in-situ measurements with the NaI detector, which provides integrated estimates of ^{137}Cs within the field of view of each measurement (3.1 m^2). There was no dependency of ^{137}Cs on pH, clay content and carbon content. However, a close relationship was determined between ^{137}Cs and soil moisture. Thus, in-situ data must be corrected for soil moisture. Close correlation ($R^2 = 0.86$) was found for ^{137}Cs activities (in Bq kg^{-1}) estimated with in-situ (NaI detector) and laboratory (GeLi detector) methods which proves the validity of the in-situ measurements with the NaI detector system. This paper describes the calibration of the NaI detector system for field application under elevated ^{137}Cs activities originating from Chernobyl fallout.

5.2 Introduction

The measurement of Caesium-137 (^{137}Cs) concentrations is often used to gain important information on the extent of soil erosion in areas where ^{137}Cs occurs either due to nuclear weapon testing and/or the Chernobyl reactor accident in 1986. After deposition ^{137}Cs is rapidly and tightly bound to soil fine particles. Redistribution is mainly caused by soil erosion where ^{137}Cs moves with soil particles (e.g. Bonnett, 1990; Ritchie & McHenry, 1990).
Measurement of ^{137}Cs can either be done in the laboratory or in-situ in the field. Advantages and disadvantages of field and laboratory applications of gamma detectors have already been discussed in detail (e.g. Beck et al., 1972; Miller & Shebell, 1993; He & Walling, 2000). Measurement time for in-situ measurements is generally shorter due to the coverage of a representative sampling area in the field which results in a better counting statistics in contrast to small volume soil samples in the laboratory (Table 5.1; Beck et al., 1972). Comparability of in-situ and laboratory measurements has been shown before, but mainly for

Ge detectors and/or on cultivated (mostly ploughed) fields with low heterogeneity (Haugen, 1992; Agnesod et al., 2001; Tyler et al., 2001). In general, because of the good resolution of peaks in gamma spectra measured with Ge detectors, these are usually favoured over the NaI detectors whose spectra often show interference of neighbouring peaks (Table 5.1). However, for application in a mountain environment with difficult accessibility and steep slopes of up to 45° priorities must be set differently from those of accessible areas or for arable lowland sites. Germanium detector systems are usually relatively heavy because of the need for cooling. In contrast to Ge detector systems, NaI detectors have the advantage that they are substantially lighter to transport, cheaper to purchase and operate (Table 5.1). However, the compromise is the reduction in spectral resolution that necessitates careful spectral processing (Table 1; Beck et al., 1972; ICRU, 1994). The aim of this study was to find a measurement routine with a NaI detector which is rather quick and can be handled by one person in the field and which still achieves accurate results.

In order to make a statement about ^{137}Cs inventories and for use as a tracer for soil erosion knowledge about spatial distribution of ^{137}Cs is crucial. The vertical distribution of ^{137}Cs in cultivated soils is influenced by the tillage practice which results in a more or less homogenous distribution of ^{137}Cs within the plough layer (Schimmack et al., 1994; Owens et al., 1996; He & Walling, 2000; Ritchie & McCarty, 2003). In unploughed soils most of the ^{137}Cs is accumulated at the top of the soil profile or few centimetres below and the content decreases exponentially with depth (Mabit et al., 2008, Ritchie & McHenry, 1990; Owens et al., 1996; Ritchie & McCarty, 2003). It is possible to distinguish areas of net soil loss from net deposition areas by analyzing the spatial distribution of ^{137}Cs in the studied area (Mabit et al., 2008, Ritchie & McHenry, 1990). Erosion investigations using ^{137}Cs from weapons testing fallout often assume that ^{137}Cs distribution is relatively homogeneous from the field to small catchment scale (e.g. Walling & Quine, 1991; Wu & Tiessen, 2002; Schoorl et al., 2004; Heckrath et al., 2005). Input of 137Cs through Chernobyl reactor accident was highly dependent on the rainfall pattern which caused high (kilometre to regional scale) heterogeneity in 137Cs distribution (Higgitt et al., 1992; Renaud et al., 2003). For a small catchment or single hillslopes a homogeneous rainfall pattern can be assumed.

There are only few studies about 137Cs in alpine regions (e.g. Hofmann et al., 1995; Albers et al., 1998; Agnesod et al., 2001) which mainly analysed soil samples in the laboratory. Albers et al. (1998) studied the distribution of ^{137}Cs in alpine soils and plants in the German Kalkalpen. They found that 10 years after the fallout from the Chernobyl reactor accident most of the ^{137}Cs was still stored in the top 5 cm of the soil profile and uptake by plants was limited.

This study aims at finding a suitable measurement routine to describe eroded and uneroded hill slopes in mountain regions and to provide a method routine for future erosion measurements. Therefore an application-oriented method to give a quick overview of ^{137}Cs distribution in the field had to be developed. In a first step small- and large-scale heterogeneity of ^{137}Cs distribution in the catchment was analysed by measuring soil samples of an altitudinal transect in the laboratory using a GeLi detector. GeLi measurements of soil samples in the laboratory were also made in order to check the comparison with ^{137}Cs specific activity concentrations and their dependency on soil parameters.

Comparison of soil sample measurements in the laboratory (GeLi detector) and in-situ measurements in the field (NaI detector) was done in order to calibrate the field device. Subsequently the calibration of the field device was validated by comparing GeLi laboratory and NaI in-situ measurements of 12 sites.

5.3 Methods and materials

5.3.1 Site

The study area is located in the southern part of Central Switzerland (Canton Uri) in the Urseren Valley (Figure 5.1). The bottom of the W-E extended mountain valley is approximately 1450 m a.s.l. (above sea level). It is surrounded by mountain ranges of altitudes up to 3200 m a.s.l.. The mean annual rainfall is 1516 mm (1986 – 2007, Source: MeteoSwiss) and the mean annual air temperature is 4.3 °C (1986 – 2007, Source: MeteoSwiss). The valley mainly consists of cultivated grasslands. Forested areas are limited to protection forests at slopes above villages. Land use is dominated by grazing and, in the lower reaches of the valley, by hay harvesting. The valley is strongly affected by soil erosion. Vegetation cover is disturbed especially on the southern slopes. Soils in the study area mainly consist of podsols and cambisols often with stagnic properties (WRB, 2006). For a detailed description of the Urseren Valley see Meusburger and Alewell (2008).

Sampling took place at 12 sites at the south exposed slope (Figure 1). Sites represent hillslope sections of about 20 m slope length. Mean slope angle of these hillslopes is 36°. For laboratory measurements soil samples of a depth of 10 cm were taken as most of the ^{137}Cs is stored in this section (e.g. Schimmack et al., 1989; Owens et al., 1996; Schoorl et al., 2004) and every site was measured three times with in-situ NaI spectrometry. For NaI system calibration 9 additional soil samples taken from a circular area of 3 m of diameter at one of the sites were used.

Additionally, an altitudinal transect between 1500 and 2050 m a.s.l. and two reference sites with high vegetation cover at sheltered positions behind geomorphologic barriers near the valley bottom (1470 m a.s.l.) were sampled (Figure 5.1). From each site 3 replicate samples were taken within 1 m^2. The reference sites represent the original ^{137}Cs activity without soil redistribution processes.

Samples were collected during summer seasons 2006 and 2007 using a core sampler of 72 mm of diameter.

5.4 Analysis

5.4.1 Laboratory measurements

Soil samples were dried at 40 °C, passed through a 2 mm sieve and finally ground using a tungsten carbide swing grinder. The ground soil samples were filled into 25 ml sample containers (6.5 cm diameter; Semadeni25) and measured for 8 hours. Measurements were done with a Li-drifted Ge detector (GeLi; Princeton Gamma-Tech, Princeton, NJ, USA) at the Department for Physics and Astronomy, University of Basel. The size of the detector is 48 mm in diameteter and 50 mm in length. The relative efficiency is 18.7% (compared to a 3 x 3 inch NaI detector at a distance of 25 cm between sample and detector). In order to reduce the amount of radiation from background sources in the environment the samples were shielded by 4 cm-thick lead during measurement. The 137Cs activity concentrations were determined using the InterWinner 5 gamma spectroscopy software (Ortec, Oak Ridge, TN, USA). The energy calibration of the GeLi detector was done using a Eu-152 multi-source with peak line positions at 117.6, 347.6, 773.5, 1108.0 and 1408.9 keV. For efficiency calibration three reference-samples provided by H. Surbeck (University of Neuchâtel) enriched with known activities of U-238, Th-232 and K-40 were used. These calibration samples were of the same geometry and a comparable density as our analysed soil samples. The resulting measurement uncertainty on ^{137}Cs peak area (at 661 keV) is lower than 15 % (error of the measurement at 2-sigma). The minimum detection activity for ^{137}Cs is 0.1 Bq kg^{-1}.

5.4.2 In-situ measurements

For field measurements a 2 x 2 inch NaI-scintillation detector (Sarad, Dresden, Germany) was used. The field equipment consists of a detector, a pole, a control unit, a battery and a pocket PC (Figure 5.2). The detector was mounted perpendicular to the ground at a height of 25 cm. Usually detectors are mounted 1 m above floor which means that the yield area has, as a rule of thumb, a radius of 10 m. For comparable results the yield area should be flat and should not have irregularities (Laedermann et al., 1998; He & Walling, 2000).

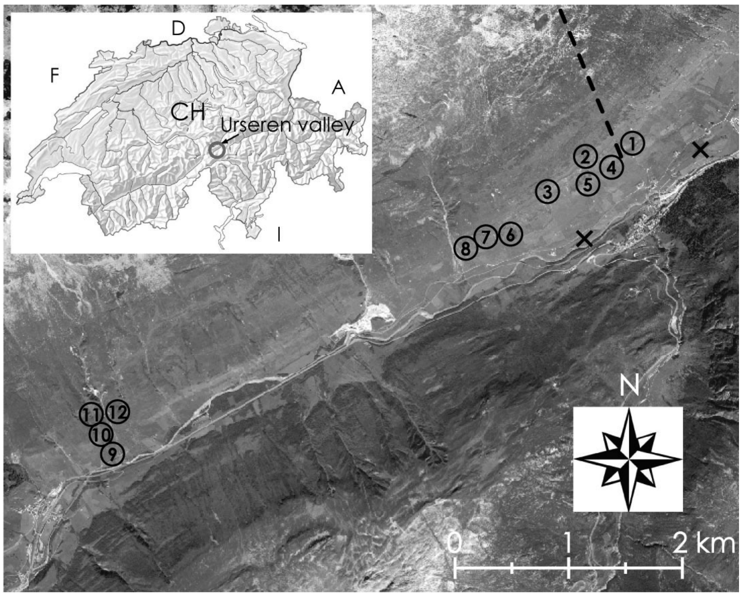

Figure 5.1: Aerial photograph of the Urseren Valley in Southern Central Switzerland and location of the sites (numbers), the altitudinal transect (dashed line) and the reference sites (crosses).

As this can not be fulfilled for such a large area at any site in a mountain region, we decided to use a detector height of 25 cm. The area of yield according to this detector height was determined (see below). For all in-situ measurements ^{137}Cs concentration was determined for 1 hour. Every hillslope section was measured three times. Measuring points were evenly distributed over the slopelength of 20 m. Spectra evaluation was done with software provided by H. Surbeck, University of Neuchâtel/Switzerland. Peak area is specified by a gauss function over the region of interest (ROI). The background is separated from the peak area by a straight line between the beginning and end point of the ROI. The ROI was set to include the three interfering peaks Tl-208 at 583.2 keV, Bi-214 at 609.3 keV and ^{137}Cs at 661.7 keV. This multiplet was deconvolved in order to get peak areas of single peaks. In areas with elevated 137Cs activities interference from natural radionuclides, full energy peaks and secondary scattering of higher energy gamma photons, are rather small. In lower activity environments a more sophisticated processing of the spectra might be required. All measured ^{137}Cs activities refer to 2007. The spectral resolution of the NaI detector is 9.7 % at 661.7 keV FWHM.

Figure 5.2: NaI detector system with its components: (a) detector, (b) pole, (c) control unit, (d) battery and (e) interface to pocket PC.

5.5 Method concepts, results & discussion

5.5.1 Spatial heterogeneity of ^{137}Cs

We tested the small-scale homogeneity of ^{137}Cs distribution in the field by using the soil samples from the altitudinal transect from 1500 m a.s.l. to 2050 m a.s.l. (Figure 5.1). Samples were taken at sites with high vegetation cover and no visible erosion and measured with the GeLi detector in the laboratory. ^{137}Cs activities of the soil samples varied between 47 and 363 Bq kg^{-1} (Figure 5.3). On single altitudinal levels a large difference in ^{137}Cs activity was measured for replicate samples within 1 m^2. Generally, the wide variation of ^{137}Cs concentration at meter scale gives evidence that either ^{137}Cs was not evenly deposited and/or not bound homogeneously to soil particles. Another possibility would be secondary processes such as erosion and accumulation, which can lead to the high small scale heterogeneity of ^{137}Cs. The problem of inhomogeneity in ^{137}Cs distribution has been shown in other studies, but mostly relating to a larger scale (e.g. Clark & Smith, 1988; Albers et al., 1998; Kaste et al., 2006; Machart et al., 2007). According to Machart et al. (2007) the distribution of the radionuclide in the contaminated cloud as well as local inhomogeneities in rainfall intensity or input on a snow layer can be the cause for inhomogeneous small-scale spatial distribution of ^{137}Cs, especially in mountainous terrain. Inhomogeneity in ^{137}Cs distribution may also be due to runoff connected to the rainfall event depositing the ^{137}Cs (Albers et al., 1998). The higher altitudes were still snow-covered at the time of ^{137}Cs deposition (Source: MeteoSchweiz). Therefore, snow melt and runoff processes must be considered as a reason for heterogeneous distribution of ^{137}Cs, also on a small scale (Haugen, 1992). Small-scale heterogeneity in ^{137}Cs activities for replicate samples at altitudes above 1800 m a.s.l. is particularly striking. There are two interpretations possible: (i) a linear trend line through all data points shows an increase of ^{137}Cs with altitude (Figure 5.3) or (ii) extreme outliers at 1850, 1950 and 2050 m a.s.l. are excluded which results in a homogeneous distribution of ^{137}Cs over 1800 m a.s.l.. The linear trend through the data points is supported by other studies that have found a dependency of ^{137}Cs on altitude (McGee et al., 1992; Arapis & Karandinos, 2004). However, interference with snow cover (Haugen, 1992) as an explanation for outliers in higher altitudinal levels is very likely. The remaining data from an altitude above 1800 m a.s.l. is similar to the reference value (146.4 ± 19.5 Bq kg^{-1}) determined near the valley bottom (Figure 5.3). Even though no erosion damage is visible in the grasslands today, the lower activities measured at sites below 1800 m a.s.l. give evidence for soil loss through erosion. This interpretation can be supported by the correspondence of lower 137Cs activities with more intensive land use practice below 1800 m a.s.l. (Meusburger and Alewell, 2008). Small scale heterogeneity in ^{137}Cs distribution is obvious from replicate samples taken at the same altitudinal level.

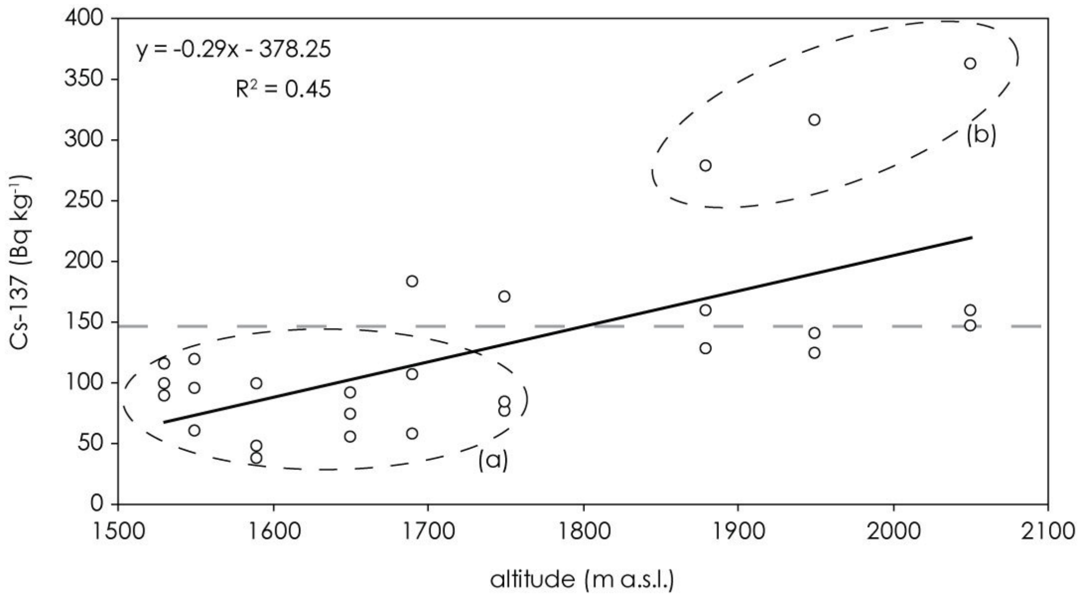

Figure 5.3: Altitudinal transect between 1500 and 2050 m a.s.l. with trendline (black). Dashed line represents the Cs-137 activity (146.5 Bq·kg^{-1}) measured at reference sites. For cluster (a) influence of erosion is possible, cluster (b) is influenced by snow at the time of Cs-137 input.

5.5.2 NaI detector calibration and boundary conditons

The ^{137}Cs peak area obtained from measurements with the NaI detector had to be converted to Bq kg^{-1}. This was done by comparison of an in-situ measurement using NaI detector to laboratory measurements using GeLi detector for the samples collected at the same site. Measurements from the altitudinal transect have shown large differences in their ^{137}Cs content in replicate samples of the same sites (Figure 5.3). Therefore, small-scale heterogeneity of ^{137}Cs concentration was analysed for NaI system calibration by laboratory measurement (GeLi detector) of 9 samples (0 - 10 cm depth) distributed over a circular area with 3 m of diameter centred on the position of the NaI in-situ measurement (Figure 5.4). Mean ^{137}Cs concentration was 58.8 ± 8.2 Bq kg^{-1} with a maximum at 68.2 Bq kg^{-1} and a minimum at 42.0 Bq kg^{-1}. The mean value of these 9 single measurements (GeLi) was used as a calibration value with which the factor for the conversion of the ^{137}Cs peak area (counts) determined from in-situ measurement (NaI) to Bq kg^{-1} was estimated. This factor allows the conversion from peak area in counts to Bq kg^{-1} for further in-situ measurements at comparable soils in the Urseren Valley. A potential dependency on soil parameters is described below.

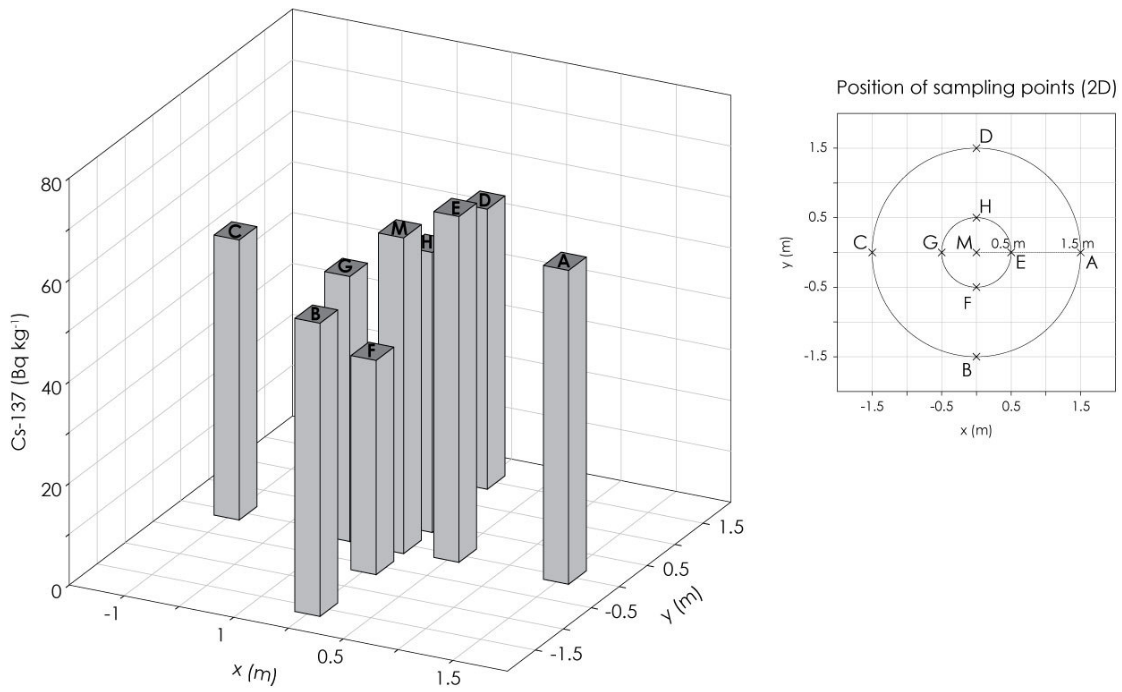

Figure 5.4: Spatial heterogeneity of the Cs-137 distribution at the calibration site.

In order to estimate the soil volume which is representative for the ^{137}Cs measured in the field, depth and spatial yield of the detector (NaI detector) as well as ^{137}Cs distribution in soil depth were determined. Depth distribution of ^{137}Cs was measured on a subdivided soil core (0 – 5 cm, 5 – 10 cm, 10 – 15 cm, 15 – 20 cm) by laboratory measurements (GeLi detector). The ^{137}Cs activity decreased exponentially with depth (Figure 5.5a). This is a typical distribution for unploughed soils (Ritchie & McHenry, 1990; Owens et al., 1996; Ritchie & McCarty, 2003). Over 70 % of the total ^{137}Cs inventory of the soil is in the top 10 cm of the soil column (Figure 5.5a). This is in accordance with Schimmack & Schultz (2006) who studied the migration of ^{137}Cs deposited after the Chernobyl reactor accident in a grassland for 15 years.

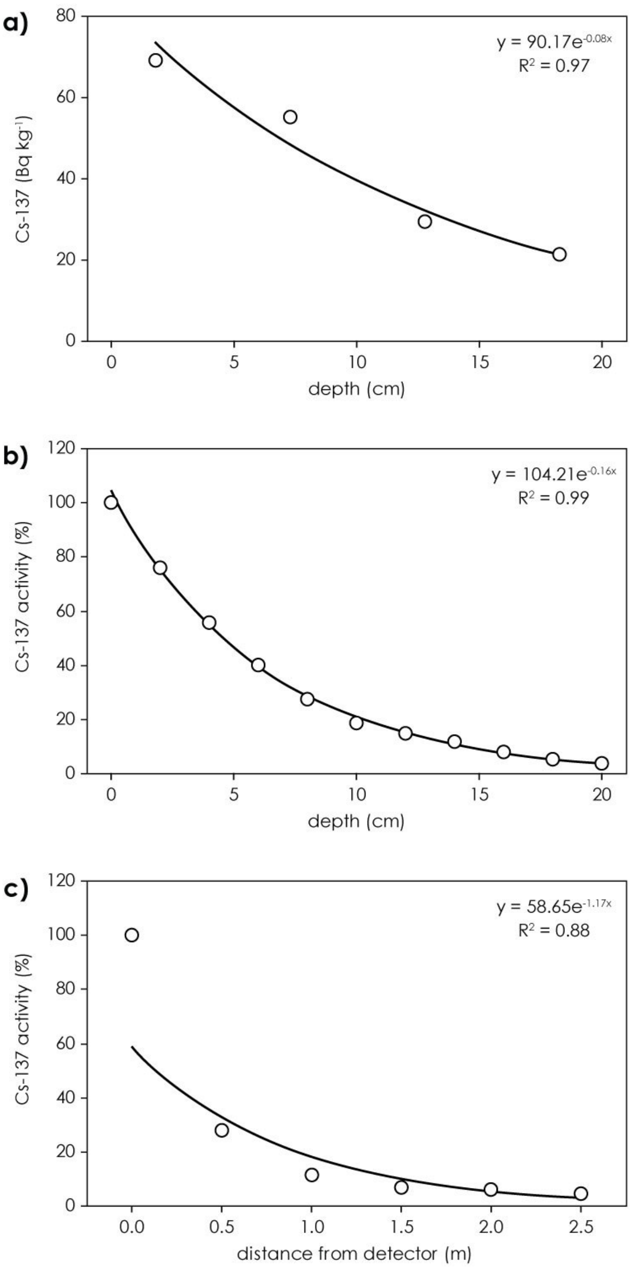

Figure 5.5: (a) Cs-137 depth profile for an upland soil measured in the laboratory (GeLi detector). (b) Exponential decrease of the Cs-137 radiation measured at the surface with increasing burial depth of the point source (NaI detector). (c) Exponential decrease of the measured Cs-137 activity with increasing horizontal distance between the point source and the detector (NaI detector).

Contribution of radiation from different soil depths to the NaI detector was estimated by repeated measurement of a buried ^{137}Cs point source (7.6 kBq 08/2007) in steps of 2 cm between 0 and 20 cm soil depth by in-situ measurement with the NaI detector mounted 25 cm above the surface. The radiation contributing to the total measured ^{137}Cs activity decreases exponentially with depth (Figure 5.5b). Most of the activity comes from the top 10 cm of the soil. Radiation from below 18 cm has a minor influence on the measured surface activity.
The horizontal coverage of the detector (NaI) was estimated in the field by repeated measurement of a ^{137}Cs point source (15.8 kBq 08/2007) with increasing horizontal distance from the detector which was mounted on 25 cm above ground. The distance was increased in 0.5 m steps. For increasing horizontal distance of the point source from the detector ^{137}Cs

activity decreases exponentially (Figure 5.5c). Under the assumption of a homogenous distribution, 90 % of the measured ^{137}Cs activity origin from a circular area with a radius of 1 m around the detector (3.1 m^2).

5.5.3 Sources of error

Determination of the ^{137}Cs peak position in the spectrum obtained by NaI measurement is subjective. Small changes in start and end position of the region of interest (ROI) leads to a big variability in peak area. This error on peak area was determined by using the mean standard deviation of peak areas of 20 test spectra each evaluated by eight persons independently. Uncertainty on peak area of measurement was 17.3 %. Interference with neighbouring peaks may lead to an uncertainty on ^{137}Cs peak area. As spectral deconvolution was processed the same way for all spectra, this uncertainty is thought to be negligible for intercomparison of samples in this study. However, for comparison of absolute values with other studies an uncertainty should be considered.

Knowledge about contribution of pre-Chernobyl ^{137}Cs is important for quantitative estimation of erosion rates because date of input is crucial to know. Contribution of pre-Chernobyl ^{137}Cs originating from above-ground nuclear weapon tests in the 1950's and 1960's is not clear for the site Urseren Valley as only little data is available for Switzerland. Riesen et al. (1999) measured samples collected in 1986 before the Chernobyl reactor accident from 12 forested sites distributed over Switzerland. The ^{137}Cs activities of the top soil layers (0 – 5 cm) were between 2 and 58 Bq kg^{-1} (Riesen et al., 1999). Consequently, after decay with a half-life of 30.17 a, in 2007 only 1 – 35 Bq kg^{-1} are left, which means that the maximum contribution of pre-Chernobyl ^{137}Cs might amount to 20 % at reference sites.

Additionally, vertical migration must be considered. From literature migration values between 0.03 and 1.30 cm a^{-1} are known (Schimmack et al., 1989; Arapis & Karandinos, 2004; Schuller et al., 2004; Schimmack & Schultz, 2006; Ajayi et al., 2007). However, our measurements show that in the Urseren Valley most of the ^{137}Cs is still stored in the top 10 cm of the soil profile. Thus, only moderate migration of ^{137}Cs occurred.

5.5.4 Dependency of ^{137}Cs on different soil parameters

Cs-137 content of the soil was controlled for dependency on pH, clay content, soil moisture and organic carbon content (Figure 5.6). There are two groups of soil differing in pH, one with a pH between 4 and 5 and one with a pH around 7 (Figure 5.6a). Neither an overall dependency of Cs-137 content on pH was found nor within the single groups. This is in accordance with Livens & Loveland (1988) who relate the pH range between 4 and 7 to conditions of immobile Cs-137. No correlation between clay content and Cs-137 content was calculated due to different sites forming different clusters (Figure 5.6b). Niesiobedzka (2000) found a positive dependency of Cs-137 on clay content for soils with very low clay contents under 3 %. Higher ^{137}Cs with higher clay content are consistent with the theory of the clay's higher binding capacity and strength. No dependency of Cs-137 on total organic carbon content was found (Figure 5.6c). Dependency of ^{137}Cs on soil moisture was estimated by repeated measurement of a soil sample at different moisture levels with the NaI detector in the laboratory. The sample was sealed in a plastic bag during measurement time to keep moisture constant. Soil moisture shows a clear influence on the measured ^{137}Cs activity (Figure 5.6d). Therefore, soil moisture was measured parallel to in-situ measurements in order to correct data with soil moisture using an EC-5 soil moisture sensor (DecagonDevices, Pullman, WA, USA). Measured soil moisture in the field at the time of in-situ measurements varies from 5 to 12 %, which represents a shielding of gamma activity by 3 to 7 %.

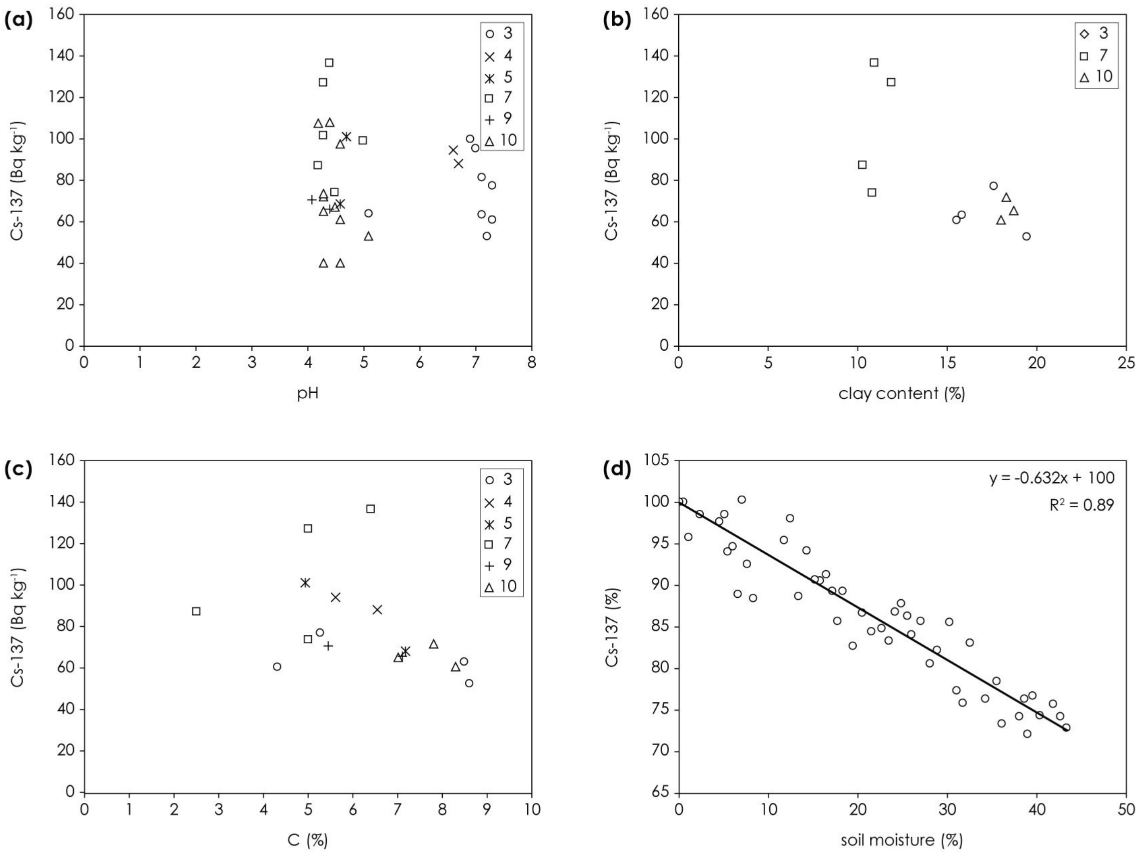

Figure 5.6: Dependency of Cs-137 on (a) pH, (b) clay content,(c) carbon content and (d) soil moisture (3, 4, 5, 7, 9, 10 stand for different sampling sites).

5.5.5 Validation of the in-situ NaI measurements

In-situ and laboratory determination of ^{137}Cs activity were compared in order to check the validity of the calibration of the NaI detector system. The mean value of all soil samples (0 – 10 cm) of one site measured in the laboratory (GeLi detector) was checked against the mean value of the in-situ measurements (NaI detector) at the same sites (Table 5.2, Figure 5.7). A good correspondence of ^{137}Cs activities determined by in-situ NaI measurements and laboratory GeLi measurements was found (R^2 = 0.86). In Figure 5.7 standard deviations for both, laboratory measurements with GeLi detector and in-situ measurement with NaI detector are given. Note that the big variation of ^{137}Cs activities for GeLi detector measurements at single sites is due to spatial heterogeneity in ^{137}Cs distribution (see above). In-situ measurements generally have a smaller standard deviation because the detector integrates over a measurement area of 3.1 m^2 (Table 5.2, Figure 5.7). The close correlation between in-situ and laboratory measurements shows that both methods give good averages of ^{137}Cs activities at alpine hillslopes. Our results are supported by Haugen (1992) and Tyler et al. (2001) who have found similar correspondences for cultivated fields.

Despite the fact that our estimations do not include an exact calculation of the basic conditions concerning ^{137}Cs flux and detector response (e.g. Beck et al., 1972; Miller &

Shebell, 1993) the achieved results with low calculating effort are equal to laboratory measurements (GeLi detector) of several soil samples. NaI in-situ measurements provide a quick and easy method to determine ^{137}Cs inventories.
Spatial small-scale distribution and especially vertical distribution of ^{137}Cs in grasslands is much more variable than in cultivated lands where ^{137}Cs is mechanically homogenised by ploughing. Therefore, interpretation of in-situ data of alpine grasslands is subject to errors relating to the spatial heterogeneity. Our data shows that ^{137}Cs varies greatly over a small scale. Consequently, either the number of soil samples per site must be increased or in-situ measurements which smooth out irregularities must be done in order to achieve a representative ^{137}Cs activity of a site. Collecting large numbers of samples in the field does not seem adapted to the case of sensitive mountain soils seriously affected by soil erosion. Especially in regions where soil restoration takes hundreds of years, non-destructive in-situ measurements should be favoured. Furthermore, large numbers of soil samples are always connected to the problem of adequate homogenizing which is problematic in densely rooted soils, soils with varying soil texture and/or organic content.

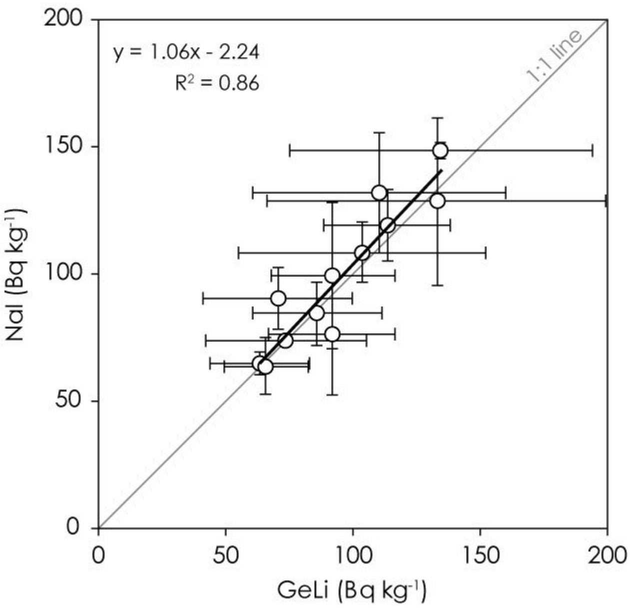

Figure 5.7: Comparison of Cs-137 activities determined by in-situ (NaI detector) and laboratory measurements (GeLi detector).

Based on the results of this study, in-situ measurements with NaI detector enable a determination of a mean ^{137}Cs activity of a single hillslope section and at the same time an estimation of the erosion state in a steep alpine catchment characterised by elevated ^{137}Cs activities originating from Chernobyl fallout. For an application of the method in the Urseren Valley see Konz et al. (2009).

Table5.1: Cs-137 data of all sites for measurements with GeLi- and NaI detector.

	GeLi				NaI			
site	Mean ($Bq \cdot kg^{-1}$)	s.d. ($Bq \cdot kg^{-1}$)	Min. ($Bq \cdot kg^{-1}$)	Max. ($Bq \cdot kg^{-1}$)	Mean ($Bq \cdot kg^{-1}$)	s.d. ($Bq \cdot kg^{-1}$)	Min. ($Bq \cdot kg^{-1}$)	Max. ($Bq \cdot kg^{-1}$)
1	73.9	31.5	42.7	134.0	73.3	0.9	72.2	73.8
2	134.8	59.4	68.6	266.0	148.0	3.1	144.4	150.5
3	104.0	48.5	52.6	177.8	108.2	12.0	95.3	119.0
4	91.9	24.9	59.59	144.5	75.8	23.2	56.2	105.9
5	92.2	24.3	48.83	132.5	99.1	28.5	70.4	135.2
6	113.8	24.9	76.8	131.2	118.6	14.1	103.0	130.5
7	133.6	67.4	53.9	260.9	128.3	32.8	104.9	165.8
8	110.8	49.8	49.6	160.3	131.7	23.8	113.7	158.6
9	66.0	16.4	40.55	84.95	63.6	10.9	52.5	63.8
10	63.7	19.3	37.7	91.4	64.4	4.6	59.2	68.2
11	86.2	25.5	55.3	112.5	84.1	12.6	70.0	94.5
12	70.7	29.3	41.0	127.8	90.1	12.2	82.6	104.1

5.6 Conclusion

The ^{137}Cs distribution in the Alps is very heterogeneous especially on a small scale (meter scale) which was shown by GeLi laboratory measurements. In alpine areas the steep terrain with low accessibility hampers the use of the heavy and mostly non portable Ge detectors in the field; hence NaI detector system offers a good alternative. Further, in-situ measurements are non-destructive which is important particularly at sites strongly affected by soil erosion. For the in -situ measurements a NaI detector was mounted 25 cm above ground integrating over a measurement area of 3.1 m^2. No dependency of Cs-137 on pH, carbon content and clay content was found, but in-situ measurements must be corrected for soil moisture. The NaI detector which is a quick and easy to apply method was successfully applied to an alpine grassland with strong heterogeneity in ^{137}Cs distribution. The use of NaI detectors in the field is a valid and rather quick alternative for extensive soil sampling (correspondence to GeLi laboratory measurements R^2 =0.86). In addition small scale heterogeneity of ^{137}Cs activity obtained in GeLi laboratory measurements is averaged spatially by NaI in-situ measurements.

Acknowledgements

This work was financially supported by the Swiss National Science Foundation (SNF), project no. 200020-113331.

We would like to thank H. Surbeck (University of Neuchâtel) for help with spectra evaluation and provision of software and samples for detector calibration, S. Estier (Federal Office of Public Health FOPH) for helpful discussion and J. Jourdan and D. Sacker (Department of Physics and Astronomy, University of Basel) for access to and help with the GeLi detector.

CHAPTER 6

Soil erosion modelled with USLE and PESERA using QuickBird derived vegetation parameters in an alpine catchment

Published in: International Journal of Applied Earth Observation and Geoinformation
Vol. 12, Issue 3, pages 208-215, 2010.
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Abstract

The focus of soil erosion research in the Alps has been in two categories: (i) on-site measurements, which are rather small scale point measurements on selected plots often constrained to irrigation experiments or (ii) off-site quantification of sediment delivery at the outlet of the catchment. Results of both categories pointed towards the importance of an intact vegetation cover to prevent soil loss. With the recent availability of high-resolution satellites such as IKONOS and QuickBird options for detecting and monitoring vegetation parameters in heterogeneous terrain have increased. The aim of this study is to evaluate the usefulness of QuickBird derived vegetation parameters in soil erosion models for alpine sites by comparison to Cesium-137 (Cs-137) derived soil erosion estimates. The study site (67km^2) is located in the Central Swiss Alps (Urseren Valley) and is characterised by scarce forest cover and strong anthropogenic influences due to grassland farming for centuries. A fractional vegetation cover (FVC) map for grassland and detailed land-cover maps are available from linear spectral unmixing and supervised classification of QuickBird imagery. The maps were introduced to the Pan–European Soil Erosion Risk Assessment (PESERA) model as well as to the Universal Soil Loss Equation (USLE). Regarding the latter model the FVC was indirectly incorporated by adapting the C factor. Both models show an increase in absolute soil erosion values when FVC is considered. In contrast to USLE and the Cs-137 soil erosion rates, PESERA estimates are low. For the USLE model also the spatial patterns improved and showed "hotspots" of high erosion of up to 16 t.ha^{-1}.a^{-1}. In conclusion field measurements of Cs-137 confirmed the improvement of soil erosion estimates using the satellite derived vegetation data.

6. 1 Introduction

Alpine areas have a high potential soil erosion risk associated to the extreme climatic and topographic conditions. The range of soil erosion rates is very uncertain due to the high spatial heterogeneity of erosion risk factors that cause difficulties to extrapolate sediment measurements on plot scale to larger regions (Helming, et al., 2005). Several models have been developed for soil loss quantification (e.g. USLE, RUSLE, LISEM, WEPP, PESERA and EROSION-3D). More recently, water erosion models tended to place a greater emphasis on representing the physical processes that are responsible for erosion, but such efforts are scarce for the alpine environment (Cernusca, et al., 1998). In the European Alps, only the Universal Soil Loss Equation (USLE) and the Pan-European Soil Erosion Risk Assessment (PESERA) model have been used (Joint Research Center Ispra, 2009a, b). USLE is an empirical model, which allows the prediction of average annual soil loss based on the product of five erosion risk factors (Wischmeier and Smith, 1978). Emerging geospatial techniques (Desmet and Govers, 1996; Reusing, et al., 2000; Wilson and Gallant, 1996) allows extending USLE to entire watersheds and rapid spatial risk assessment, particularly for remote rural areas, where data availability is constrained. PESERA is a simple physically based model with low data demand based on a runoff threshold, which depends on soil properties, the surface and the vegetation cover. The validity of these models has to be carefully considered for alpine regions (Van Rompaey, et al., 2003a; Van Rompaey, et al., 2003b), especially for the empirical USLE model, which was designed for the western U.S.. In high relief regions with rugged topography a more detailed scale is needed. Jetten et al. (2003) even stated that there might be more benefit for soil erosion assessment by improving spatial information for model input and validation rather than by adapting models to a specific landscape. Satellite imagery can provide valuable spatial information mainly on vegetation parameters to improve the performance of soil erosion models (De Asis and Omasa, 2007; De Asis, et al., 2008; De Jong, 1994; De Jong, et al., 1999; Jain, et al., 2002; Tweddales, et al., 2000). PESERA considers different vegetation- management types and fractional vegetation cover. In USLE, the C factor accounts for vegetation characteristics. The C factor is an empirical factor that is dependent on vegetation type, management and fractional vegetation cover. Several studies focused on the mapping of vegetation parameters using techniques like image classification, NDVI and linear spectral unmixing (De Jong, 1994; De Jong, et al., 1999; Liu, et al., 2004; Thiam, 2003). The availability of high-resolution satellites such as IKONOS and QuickBird further increased the options for mapping of vegetation parameters.

This paper explores how high resolution maps of fractional vegetation cover (FVC) and land cover improve soil erosion risk mapping using USLE and PESERA in an alpine catchment. High resolution maps of land-cover and FVC were obtained from image classification and linear spectral unmixing analysis (Meusburger, et al., submitted, this volume). The evaluation of model performance is done by comparing modelled soil erosion estimates to Cesium-137 (Cs-137) measurements. Cs-137 inventories are an established approach to gain integrative soil erosion estimates (He and Walling, 2000; Ritchie and McHenry, 1990; Walling, et al., 1999).

6. 2 Site description

The study site (67 km^2) is located in the Urseren Valley (46.36°N, – 8.32°E) Central Switzerland Alps. The U-shaped valley is characterized by distinct topography with elevation ranging from 1400 to 3200 m a.s.l and a mean slope angle of 24.6°. The valley corresponds to a geological fault line that extends NW-SE. The different geologic formations are displayed in Figure 6.1. The climate is alpine with a mean air temperature of 3.1°C and a mean annual rainfall of about 1400 mm per year at the climate station in Andermatt (1901-1961; 1442 m a.s.l.) of MeteoSwiss. The valley is snow covered for 5 to 6 month (from November to April) with the maximum snow height in March. The river Reuss drains the valley. Its nivo-glacial runoff regime is replenished by summer and early autumn floods. The peak runoff period is in June (BAFU, 2009). The valley mainly consists of cultivated grasslands. Forested areas are limited to protection forests at slopes above villages. Land use is dominated by grazing and, in the lower reaches of the valley, by hay harvesting. Dominant soil types in the catchment classified after WRB (2006) are Podsols, Podzocambisols and Cambisols, often with stagnic properties. On steep valley slopes, Leptosols are common (with rendzic Leptosols on the calcareous substrates). At the valley bottom and lower slopes, clayey gleyic Cambisol, partly stagnic Histosols, Fluvisols and Gleysols developed. The predominant soil texture in the upper 10 cm is silty-loamy sand. The highest soil erosion rates are expected in spring due to the snowmelt on scarce vegetation cover. The dominant soil erosion process is sheet erosion. Rill erosion is of minor importance at our study site. For a more detailed description of the study site, see Meusburger and Alewell (2008).

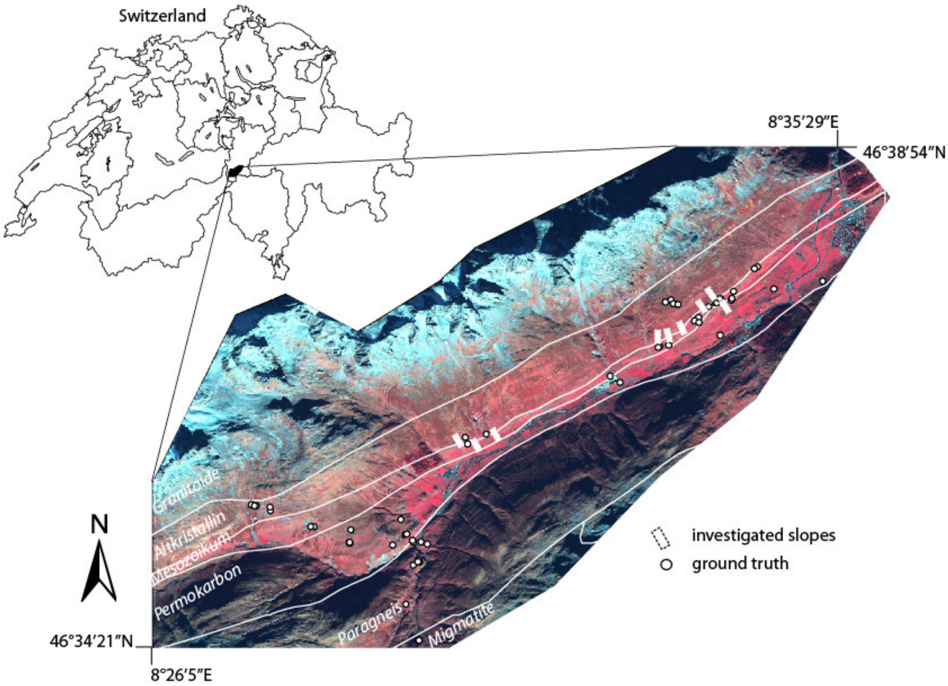

Figure. 6.1: Geographic location of the QuickBird image (false colour). The white lines separate different geologic formations, the white dots show the locations of the Cs-137 elevation transect and the white rectangles are the slopes used for the model evaluation.

6. 3 Materials and Methods

In order to compare model performance of PESERA and USLE at our alpine study site we applied the models at nine slopes (average slope angle of 37°) with measured long-term soil erosion rates based on Cs-137 method. We evaluated four different model runs: 1) run1 was done with ground truth FVC of the slopes, 2) run2 with 100% FVC, 3) run3 with 0% FVC and 4) run4 with FVC derived form linear spectral unmixing (LSU). For run4, the average FVC of the 7 pixels that match the location of each slope was used.
Further the USLE was applied for the entire catchment using the high resolution (2.4 m) FVC- and land-cover map in comparison to low resolution (25 m) land-cover data.

6. 3.1 Model input data

Parameters for the soil erosion model PESERA

The PESERA model (Kirkby, et al., 2003; Kirkby, et al., 2000) estimates soil erosion rates by the processes of sheet wash and rill wash on a regional scale. The spatially distributed model is process based and commonly applied as a diagnostic tool for land use and climate change scenarios on an European scale (Gobin, et al., 2004). The PESERA works with a regular grid and may be applied to variable scales. As data resolution becomes finer (< 100 m) and for single slopes the PESERA-VBA model is recommended and was used in this study (Joint Research Center Ispra, 2009a). The key concept of the model is the separation of precipitation into overland flow and infiltration with a runoff threshold. The runoff threshold value accounts for the effects of surface storage, the dynamic evolution of soil crusting as well as moisture storage within the upper soil layers. In addition to the runoff threshold, two further components determine the erosion estimates: erosion potential and erodibility (Kirkby, et al., 2003). Allowance is made for snow accumulation, melting and frozen ground development, using a set of accumulated day-degree terms based on daily temperature data (mean, maximum and minimum). Finally, sediment transport is calculated as:

$$S = k\, q^2\, G \tag{6.1}$$

where k is the soil erodibility, q is the overland flow discharge per unit width and G is local slope gradient. For more details on the model concept see Kirkby et al. (2003). The climate input parameters of the model are incorporated on a monthly basis for the period of 1986 to 2007 (corresponding to the time-frame of the Cs-137 method). The monthly outputs were averaged for the according period. The potential evaporation was calculated based on daily mean values of relative humidity, air temperature, global radiation and wind speed using the Penman equation (Penman, 1948). PESERA-VBA offers five soil type categories. We chose medium soil texture (18% < clay < 35% and >= 15% sand, or 18% <clay and 15% < sand < 65%) for all our sites. The topography of the single slopes was described by slope angle and curvature. Land use type was set to pasture considering the respective FVC. The ground truth FVC was determined in September 2007 with a mesh of 1 m^2 (grid size of 0.1 m^2). The FVC of each mesh was visually estimated and averaged for the entire square meter. This procedure was repeated four times for each slope. The maximum standard deviation was ~5 % (Konz, et al., 2009b).

Parameters for the soil erosion model USLE

The USLE is given as:

$$A = R \times K \times LS \times C \times P \tag{6.2}$$

where A is the predicted average annual soil loss (t ha^{-1} a^{-1}). R is rainfall- runoff-erosivity factor (N h^{-1}) that quantifies the effects of raindrop impact and reflects the rate of runoff likely to be associated with the rain (Wischmeier and Smith, 1978). The soil erodibility factor K (t ha h MJ^{-1} ha^{-1} mm^{-1}) reflects the ease with which the soil is detached by impact of a splash or surface flow. The parameter LS (dimensionless) accounts for the effect of slope length (L) and slope gradient (S) on soil erosion. The C factor is the cover factor, which represents the effects of all interrelated cover and management variables (Renard, et al., 1997). Published values of C (dimensionless) vary from 0.0005 for forest areas with 100% ground cover to 1 for bare soil areas (US Department of Agriculture, 1977). The P factor (dimensionless) is the support practice factor (Wischmeier and Smith, 1978), which is set to 1 because there are no erosion control practices in the study site. The *R, K, LS* factor basically determine the erosion volume while the C and P factor are reduction factors ranging between 0 and 1.

Typically, rainfall erosivity (R) is computed as total storm energy (MJ m^{-2}) multiplied by the maximum 30-min intensity (mm h^{-1}) (Renard, et al., 1997). Detailed data on storm intensity is available only for a limited time span (2006-2008), necessitating the use of approximation methods. We used the algorithm of Rogler and Schwertmann (1981) for the determination of the R factor, because the equation was designed for a comparable geographical and meteorological situation in the Bavarian Alps. Rogler and Schwertmann (1981) calculate R as:

$$R_r = 0.083 \times N - 1.77 \qquad 6.3$$

where R_r is the rainfall-runoff factor (kg m s^{-2} h^{-1}) and N the average annual rainfall (mm). In the Alps some of the precipitation falls as snow, which reduces R factor values. According to Schüpp (1975) the relation between elevation (m a.s.l.) and proportion of snow is:

$$f(x) = 0.0264 \times elevation - 2.0663 \qquad 6.4$$

The combined equation for the R factor is:

$$R = 0.083 \times N \times (1 - (0.0264 \times elevation - 2.0663)/100) - 1.77 \qquad 6.5$$

The elevation map was derived from the DEM (25 m grid; vertical accuracy in the Alps of ±3 m; (Swisstopo, 2006)). The mean annual precipitation map was derived through inverse distance weighted interpolation of 1 km grid values of the HADES (©Hydrological Atlas of Switzerland, Swiss Federal Office for the Environment).
The soil erodibility factor K (t ha MJ^{-1} mm^{-1}) was calculated according to Wischmeier and Smith (1978). The calculation of the K factor requires four parameters:

$$K = \left(27.66 \times m^{1.14} \times 10^{-8} \times (12 - a)\right) + \left(0.0043 \times (b - 2)\right) + \left(0.0033 \times (c - 3)\right) \qquad 6.6$$

in which m = (silt (%) + very fine sand (%) x (100-clay (%)), a = organic matter (%), b = structure code: (1) very structured or particulate, (2) fairly structured, (3) slightly structured and (4) solid and c = profile permeability code: (1) rapid, (2) moderate to rapid, (3) moderate, (4) moderate to slow, (5) slow and (6) very slow. Grain size analysis of the upper 10 cm was done for 52 samples at 18 locations with a sedigraph (5100 micromeritics). The organic matter map was derived from measurement of total carbon content at 23 soil profiles with a depth of 40 cm with a Leco CHN analyzer 1000. Measurement reproducibility was better than 0.1%. For the determination of the K factor the values of the upper 10 cm are used.
The C factor was parameterised by assigning a uniform value to each land-cover class. For grassland areas parameterisation was done according to the description of the US Department of Agriculture (1977). A more detailed description is presented in section 6.3.2.
The LS factor (dimensionless) was calculated with the DEM according to the procedure described in Renard (1997) because it is also valid for steep slopes between 22 % (12.4 °) and 56 % (29.3 °).
In addition, to the application of USLE for the nine slope sections, a parameterisation was done for the entire catchment. Therefore the point measurements needed to be regionalised.

Regionalisation of the USLE parameters to catchment scale

For the K factor, spatial estimates of grain size distribution were derived by assigning the mean value of soil samples within a geologic formation to the entire geologic formation. The geologic map was created based on the definition of geologic formations by Labhart (1999) that was refined by field and air photograph mapping. Using ANOVA we found that organic matter content significantly differs between upland and wetland soils. Thus, we mapped the distribution of organic matter via the occurrence of wetland areas. The latter was done by selecting all areas with a topographic wetness index > 50 and a slope angle < 8°. The wetland map was verified by comparison to an orthophoto of the study site. Soils above 2000 m a.s.l. were assumed to exceed organic matter contents of 12% as was found for soil samples in the study site as well as for other alpine regions (Egli, et al., 2005; Leifeld, et al., 2008). The soil structure parameter map consisted of two categories: for the Mesozoic formation with clayey soils, we set a class value of 3 (lumpy) for all other areas, we used the value 1 (granular). For the profile permeability, we used three categories 1 (rapid) for the alluvium at the valley bottom and debris fans, 5 (slow) for the Mesozoic formation and (2) moderate to rapid for the remaining areas. Structure code and profile permeability code were regionalized with the geomorphology and geology map. The geomorphologic map was generated based on a Quaternary map with a scale of 1:33000 (Fehr, 1926) and the orthophoto. All data layers were created using ArcGIS (ESRI) version 9.2.

6.3.2 QuickBird data for vegetation parameterisation

High resolution multispectral QuickBird imagery (pixel resolution = 2.4 m) was used to improve the parameterisation of vegetation. QuickBird standard imagery was acquired on October 17, 2006 (10:51 UTC), under clear sky conditions, over a 67 km^2 area. The imagery was radiometrically- and geo-corrected by the satellite data providers with a published spatial accuracy of 14 m root mean square error (RMSE). Orographic illumination effects were corrected using a DEM (25 m grid; ±3 m vertical accuracy in the Alps) and the TOPOCOR module of the Atcor3 software package (Richter, 2005).

Two datasets relevant for vegetation parameterisation result from the analysis of QuickBird imagery, a land-cover map and a FVC map. The land-cover map consists of nine categories: forest, shrub, dwarf-shrub, grassland, non-photosynthetic vegetation, snow, water, bare soil, rock and the artificial category shadow. The overall classification accuracy is 93.3%. In total, 0.81% of the dataset is unclassified. The land-cover classes of bare soil, grassland, rock, snow, water, forest, and shadow have high classification accuracies (82%~100%), while classes of dwarf-shrub, shrub, and non-photosynthetic vegetation have lower classification accuracies (67%~78%).

The FVC map for the grassland areas was generated by linear spectral unmixing (LSU) of the grassland areas using the endmembers bare soil, grassland and shadow. The resultant vegetation abundance showed significant correlation to ground truth (GT) FVC (Figure 6.2). The regression between the 43 ground truth FVC and QuickBird derived vegetation abundance points is given by:

$$FVC(GT) = 1.63 \times FVC(LSU) + 12.20 \qquad 6.7$$

with a coefficient of determination of $r^2 = 0.85$. The mean RMSE value for the image is 0.52. For a detailed description of the analysis see Meusburger et al. (submitted, this volume).

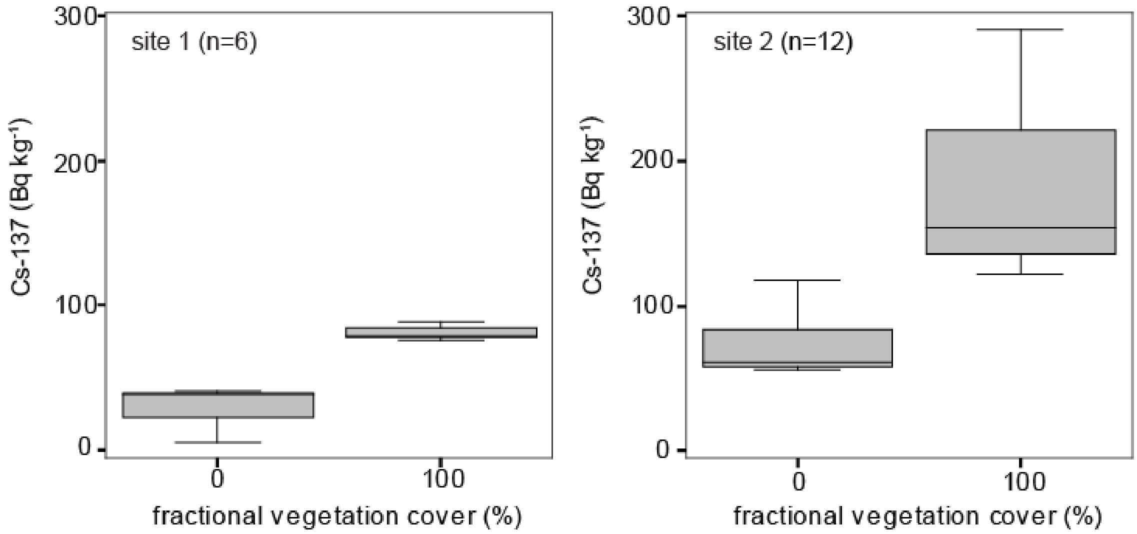

Figure 6.2: Scatter plot of LSU-derived vegetation abundances versus the ground truth fractional vegetation cover. Linear regression line is shown.

Except for the land-cover class grassland uniform *C* factors are used for single land-cover classes derived from the QuickBird classification. For the land-cover class "grassland", the following equation was used to implement the FVC (US Department of Agriculture, 1977):

$$C = 0.45 \times e^{-0.0456 \times cv} \quad 6.8$$

where cv is the "cover that contacts the surface". We assume that for grassland cv equals FVC. We will refer to this map as LSU-derived *C* factor map in the following. The resulting *C* factors are shown in Table 6.1 together with the values of the low resolution dataset.

For the application of USLE on catchment scale, comparative analysis of soil erosion estimates between the low resolution land-cover input data (uniform *C* factors are used for single land-cover classes) and high resolution satellite-derived land-cover data (considering FVC for grassland areas) was conducted (see below). The land-cover dataset (where we will refer to as the "low resolution dataset") is based on the VECTOR25 dataset available at Swisstopo with a scale of 1:25000. We used a raster-based approach for the modelling of soil erosion with a pixel resolution of 25 m for the low resolution dataset and a pixel resolution of 2.4 m equal to the spatial resolution of the QuickBird image.

Table 6.1: *C* factors used for specific land-cover classes (US Department of Agriculture, 1977) for the low resolution land-cover dataset and the QuickBird derived data. * indicates that the according land-cover class is not existent, thus, the *C* factors for grassland were assigned.

land-cover type	*C* factor for the low resolution dataset	*C* factor
bare soil	* 0.002	0.45
dwarf-shrub	* 0.002	0.011
shrub	0.003	0.003
grassland	0.002	depending on FVC
forest	0.0005	0.0005
rock, snow, non-photosynthetic vegetation	0	0

6.3.3 Cs-137 measurements

Soil erosion rates estimated with PESERA-VBA and USLE were compared to erosion rates derived from in-situ measurements of Cs-137 with a sodium-iodide (NaI) detector 2 x 2 inch (Sarad, Germany; efficiency at 661 keV is 1.4 $s^{-1}/(\mu Sv\ h^{-1})$). The detector was mounted perpendicular to the ground on a height of 25 cm. Cs-137 concentration was measured for 1 hour with three replicates per slope section. Spectra evaluation was done with software provided by H. Surbeck, University of Neuchâtel (Switzerland). All measured Cs-137 activities refer to 2007. The mean error for the erosion estimates resulting from analytics and data evaluation is ± 30% (Konz, et al., 2009b).

To evaluate the USLE estimates for the catchment seven additional Cs-137 measurements (with 3 replicates) along an elevation gradient were done. Here, only the Cs-137 activity of soil samples (upper 10cm) was determined with a Li-drifted Ge-detector and InterWinner5 gamma spectroscopy software (Schaub, et al., submitted). Spearman correlation was used to compare modelled erosion estimates with the Cs-137 activity. For all statistic analysis the software SPSS was used (SPSS, 2009). A list of the used data is presented in Table 6. 2.

Table 6.2: Used datasets (*npv = non-photosynthetic vegetation).

Data Type	Model parameters	Source	Samples (n)
Topography	LS factor map (USLE: plot, catchment) Slope angle (PESERA-VBA)	DEM ©swisstopo	-
Climate	R factor map (USLE: plot, catchment) Monthly climate data: precipitation, temperature, pot. evaporation (PESERA-VBA)	DEM, mean precipitation data ©Hydrological Atlas of Switzerland Daily climate data of MeteoSwiss:	-
Soil	K factor (USLE: plot)	Plot grain size analysis Plot organic matter content	9 (3-5 replicates) 9 (3 replicates)
	K factor map (USLE: catchment)	DEM (topographic wetness index, slope, elevation) Grain size analysis Organic matter content	9 (composite sample) 14 (5 replicates)
Vegetation	C factor map (USLE: catchment)	VECTOR25 dataset ©swisstopo	-
	Land-cover map (USLE: catchment)	QuickBird imagery	-
	LSU FVC map (USLE: plot, catchment, PESERA-VBA)	QuickBird imagery	-
	Ground truth FVC (USLE: plot, PESERA-VBA)	field measurement	9 (4 replicates)
Evaluation	Cs-137 erosion estimates (USLE: plot, PESERA-VBA)	in-situ measurement	9 (3 replicates)
	Cs-137 activities (USLE: catchment)	laboratory measurement	7 (3 replicates)

6. 4 Results and Discussion

6.4.1 Comparison of USLE and PESERA-VBA at plot scale

The mean erosion estimates for the nine plots and the four test runs using PESERA-VBA and USLE are shown in Table 6.3 (column 3-6) together with the soil erosion estimates from in-situ Cs-137 measurements (column 2). PESERA-VBA distinguishes only between 6 soil texture classes and 12 land use types, which is limiting the spatial differentiation. This is especially true for our sites that differ in soil texture, land use and FVC, of which only differences in FVC could be considered. Thus, we will only compare mean values of the nine slopes. The mean erosion rate resulting from in-situ Cs-137 measurements is 20.1 ± 5.8 t ha^{-1} a^{-1}. Both models underestimate the Cs-137 derived soil erosion estimates in all runs. With field measured data (column 3), the USLE estimate is 9.6 t ha^{-1} a^{-1}, which is only half of Cs-137 based erosion rates. PESERA-VBA computed a soil erosion rate of 0.37 t ha^{-1} a^{-1}, which is approximately 50 times less. PESERA-VBA estimates of erosion rates are far too low even with 0% FVC (column 5). Using 100% FVC (column 4), which is done in most soil erosion modelling studies, both models underestimate erosion rates modelled with field measured input data by almost a magnitude (an underestimation of 8.2 t ha^{-1} a^{-1} (85%) and 0.05 t ha^{-1} a^{-1} (86%) for the USLE and the PESERA-VBA, respectively). By introducing the LSU-derived FVC (column 6), the modelled erosion rates of both models approximate the modelled values based on the field measured vegetation cover but still underestimate Cs-137 derived soil erosion rates.

Table 6.3: Comparison between mean (nine slopes) Cs-137-derived soil erosion rates and mean estimates (t ha^{-1} a^{-1}) of USLE and PESERA for four different model runs with (1) field measured input data at plot scale (2) 100% FVC (3) 0% FVC and (4) LSU-derived FVC.

Model	Cs-137 erosion estimates	1) Measured data	2) 100% FVC	3) 0% FVC	4) LSU-derived FVC
USLE	20.1 ± 5.8	9.6	1.4	201.6	11.4
PESERA-VBA	20.1 ± 5.8	0.37	0.05	2.9	0.38

The deviation between the Cs-137 and modelled soil erosion values is high. USLE does not account for winter processes such as snow gliding, snow ablation, avalanches and snow melt. The latter processes have been discussed as potentially being the crucial confounding factors for soil erosion in the Urseren Valley (Konz, et al., 2009a). Surprisingly the deviation is even higher for PESERA-VBA, although snow accumulation and snowmelt are considered and peak erosion occurs in May during snow melt. The main reason might be the lower sensitivity of PESERA-VBA to changes in FVC, which results in a very small range of erosion estimates between 0.05 t ha^{-1} a^{-1} for 0% FVC and 2.9 t ha^{-1} a^{-1} for 100% FVC. Sprinkling experiments at the sites have shown that 100% FVC almost prevents soil erosion (Merz, et al., 2009), which is in accordance with PESERA-VBA. For 0% FVC, on sites with silty-loamy sands and an average slope angle of 37°, higher soil erosion rates are expected. Another possible reason for the underestimation of soil erosion rates for sites with low FVC might be a low erosivity resulting from monthly averaged climate input data in the PESERA-VBA that smoothes out extreme events. Even though a coefficient of variation for the monthly precipitation is considered its impact on soil erosion estimates is low. The empirical USLE equation 6.8 where the impact of FVC is implemented via the *C* factor seems more suitable than the vegetation growing module of PESERA-VBA at our site. An exponential relationship (r^2= 0.93), similar to the empirical USLE equation, between sediment yield and FVC was found for sprinkling experiments in the valley (Schindler, et al., submitted). Moreover, the empirical factors of the USLE equation are based on long-term average values (including extreme events) and the used R and LS factor are adapted to mountainous areas which might explain the higher soil erosion estimates. In general, USLE estimates are closer to the range of measured erosion rates by the Cs-137 method and to visually observed erosion features. Further it showed a high dependency on FVC, which is a decisive parameter in alpine regions (Isselin-Nondedeu and Bedecarrats, 2007).

6.4.2 USLE erosion estimates for the entire catchment

Grain size analysis for the nine slope sections (with 3-5 replicates each) showed no significant differences between slopes. Mean values are 37.86% ± 1.10 sand, 48.25% ± 1.18 silt and 13.09% ± 0.41 clay. Total carbon content of upland soils is 3.49% ± 0.17 (n = 121) and 24.85% ± 1.48 (n = 107) for wetland soils (note that most wetland soils are influenced by sedimentation of mineral soil particles thus decreasing organic matter content). For the spatial mapping of organic matter content, we worked with two groups: 24.9% for wetlands and 3.5% for the remaining areas. The produced input parameter maps for USLE are displayed in Fig. 6.3.

K factor values range from 0.0 to 0.368 t ha MJ^{-1} mm^{-1} with the lowest values in elevated areas where skeleton- and organic matter content is high. The values for *R* range from 24.4 to 84.2 N h^{-1} with the lowest values for elevated areas due to the increasing proportion of snowfall. The *LS* factor map clearly displays the channel network, where the highest values of about 57.5 are observed in the channels. The LSU-derived *C* factors are highest for bare soil areas, which are mainly located at the south exposed slope and low for rock outcrops that have no erosion potential (see enlargement in Figure 6.3).

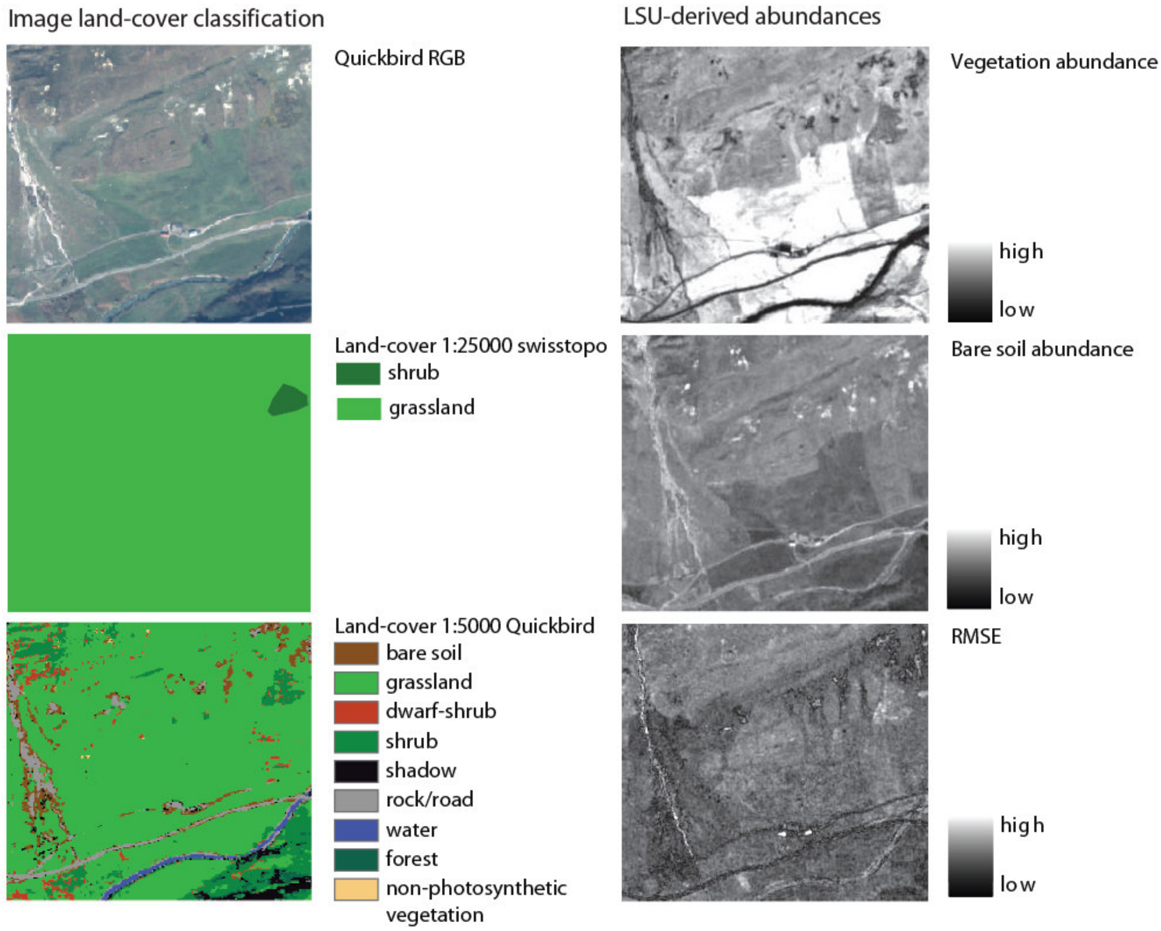

Figure 6.3: USLE input factor maps (*LS*-, *R*-, *K* factor and LSU- derived *C* factor) and an enlargement for the LSU-derived *C* factor.

The predicted soil erosion estimates are categorized into five classes (Figure 6.4). The land-cover types shadow, snow and rock are displayed in black. With the *C* factor based on the low resolution dataset highest soil erosion values are about 2.8 t ha^{-1} a^{-1} and the mean modelled erosion rate for the entire catchment is 0.08 t ha^{-1} a^{-1}. Highest values are found along the channels in the catchment. The areas in between are characterised by very low erosion rates < 0.5 t ha^{-1} a^{-1}. Thus, the former soil erosion map mainly displays the pattern of the *LS* factor map. The erosion estimates produced with the LSU-derived *C* factor map range from 0.0 to 16 t ha^{-1} a^{-1} with a mean erosion of 1.18 t ha^{-1} a^{-1}. High erosion rates are concentrated on the south exposed slopes where grassland cover is scarce due to pasturing and at landslides (Meusburger and Alewell, 2009). The highest values occur at spots of bare soil located along rivers, roads and landslides (Figure 6.4, lower part). Thus, by visual evaluation of the produced maps the satellite data assisted map is plausible.

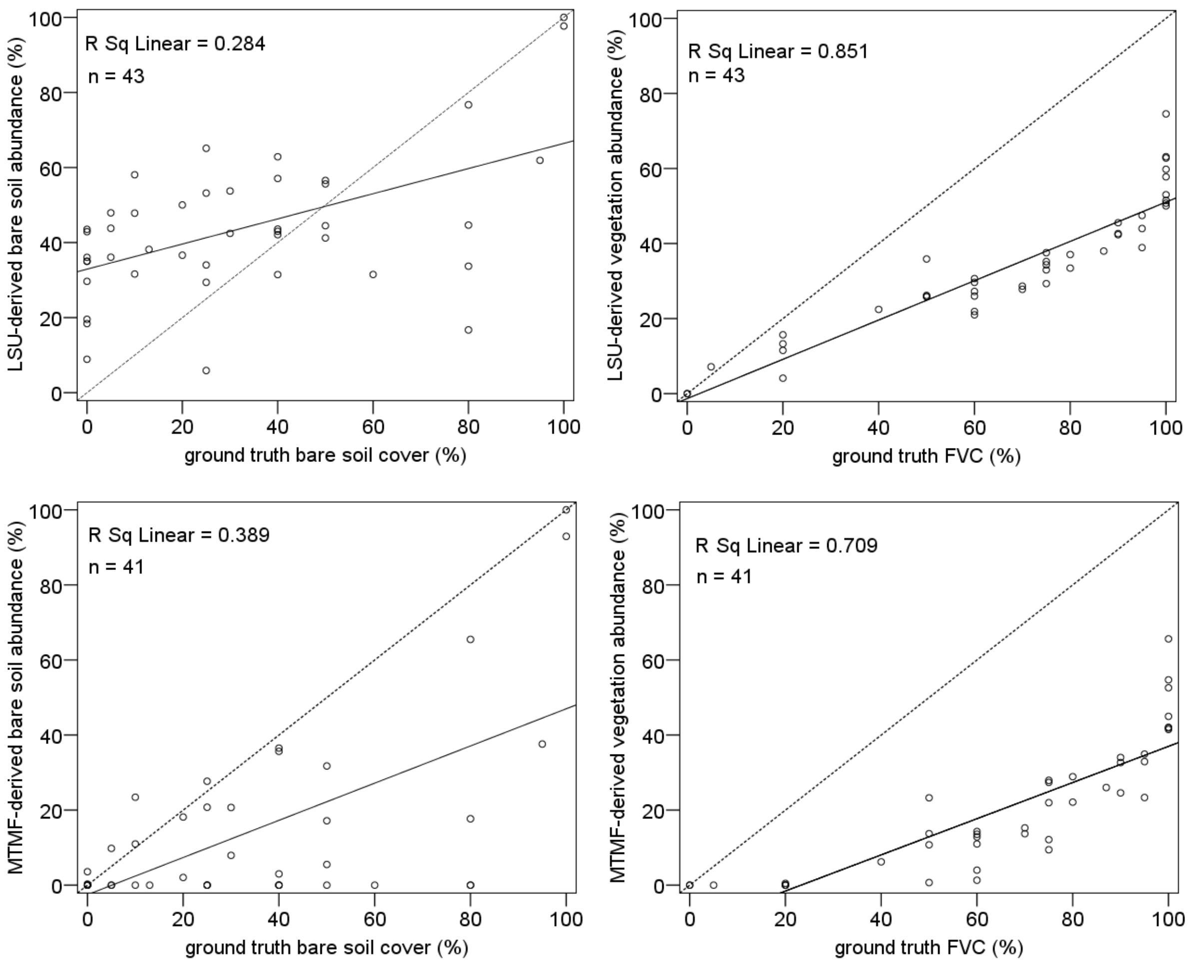

Figure 6.4: Estimated soil loss (t ha^{-1} a^{-1}) by USLE calculated with the low resolution *C* factor map (based on the swisstopo land-cover dataset) and the *C* factor based on the QuickBird derived vegetation data. Below: a visible comparison between the high resolved USLE soil erosion map and the pan-sharpened QuickBird image.

In addition, model estimates are compared to Cs-137 activities for seven locations. The Spearman's rank correlation between LSU-assisted USLE outputs and Cs-137 activity is $r = -0.71$ ($n = 7$). For the USLE map based on the low resolution dataset a correlation coefficient of $r = -0.43$ ($n = 7$) is observed. The correlation is negative because the Cs-137 activity is higher with less soil erosion (Sawhney, 1972). Both correlations are not significant due to the small number of sampled locations ($n = 7$ with 3 replicates).
Generally the evaluation with Cs-137 measurements showed that using the LSU-derived *C* factor increased the correspondence between measured Cs-137 activity and modelled values of soil erosion. Both Cs-137 and USLE give long-term erosion estimates, which makes the combination of the two methods very suitable. Furthermore, the high resolution input data allows for allocation of field measurements and observed erosion features. This is a necessary requirement in order to validate produced erosion estimates.

6.5 Conclusion and outlook

Even though USLE is an empirically based model and was not designed for the application in alpine environments the magnitude of erosion estimates seem more plausible than PESERA-VBA derived rates. This might be due to the suitability and high impact of the *C* factor to overall model outputs at our sites. Several other erosion models also apply this *C* factor such as the Morgan and Finney method (Morgan, et al., 1984), WEPP (NSERL, 1995), ANSWERS (Beasley, et al., 1980), SEMMED (De Jong and Riezebos, 1997) and PCARES (Paningbatan, 2001). Thus, improved C factor mapping using remote sensing can provide crucial information to improve spatial soil erosion modelling. PESERA-VBA under estimate the importance and influence of vegetation cover on soil erosion in alpine grasslands.

The QuickBird data could decisively improve vegetation input data for soil erosion risk mapping. On catchment scale (using USLE and high resolution vegetation data), "hotspots" of soil erosion were identified, which allows for more effective soil conservation planning. In addition to direct visual verification of the high resolution USLE map a comparison to soil erosion measurements in the field is possible, which is of crucial importance to validate and adapt soil erosion models. We conclude that even a "simple" empirical erosion model can strongly benefit from using spatial FVC information.

Generally, the suggested approach to model soil erosion with high resolution input data on the catchment scale seems to be promising especially with the perspective to validate the obtained soil erosion risk maps. Model validation is not only critical with respect to the spatial patterns of erosion estimates but also with respect to absolute erosion rates. Consequently, the main profit of high resolution data for soil erosion assessment lies in the ability to directly compare results to ground measurements while at the same time providing soil erosion estimates for entire catchments.

Acknowledgement

This study was funded by the Swiss Federal Office for the Environment (Contract-no.: StoBoBio/810.3129.004/05/0X).

CHAPTER 7

Final remarks & Outlook

Soil erosion is a serious problem in mountain ecosystems. For the purpose of a better understanding of soil degradation processes adapted approaches for alpine environments are needed to design control strategies of soil erosion in the future. In this study we investigated the suitability of measurement tools that have been applied in low land regions for tracing soil erosion.

7.1 Methodological approach

Cs-137 gamma spectrometry is commonly used for soil erosion measurements but mostly in lowlands and with laboratory measurements. In order to adapt the method to mountainous environments an in-situ NaI detector was tested for its suitability for Cs-137 measurements at steep alpine slopes. Our comparison of Cs-137 measurements from the laboratory and in-situ measurements at the same site in the Urseren Valley shows that despite the big heterogeneity in Cs-137 distribution both methods achieve similar mean Cs-137 activities (R^2 of 0.94). In-situ measurements seem to have a better suitability for erosion application in high alpine environments as an extensive soil sampling is not necessary and the spatial heterogeneity is captured. Although, Ge detector systems offer a better resolution of energy spectra priorities must be set different for sensitive alpine ecosystems. Also, soil sampling as well as sample measurements is time consuming and expensive. Thus, in-situ Cs-137 measurements are not destructive and can provide reference data to evaluate soil erosion models for long term prediction since 1986.

7.2 Erosion measurements

We tested the suitability of measurement methods of sediment traps (plot measurement), sediment cups (point measurement) and Cs-137 for tracing soil erosion in alpine environments. All studied measurement methods are common methods to quantify soil erosion rates on arable land in lowlands or low mountain regions. During summer time, sediment traps can be used to measure erosion rates. But the spatial heterogeneity can not be captured as well as the probability that the trap is on the right place to measure extreme events is low. During winter time, sediment traps turned out to be not a useful tool for soil erosion measurement. Steel plates of the traps and upper plot boundaries were flattened by the winter processes. Thus, detached soil could not be collected by the traps.

Sediment cups are a practicable tool to determine small scale erosion activity. Without upper and side boundaries, the method is qualitative. Because sediment cups are at ground level they also useful during winter time. None of our 90 cups that have been installed were destroyed during winter time. Furthermore, relative differences of soil erosion activity between different land use types can be shown. Thus, soil erosion activity on point turned out to be a practicable method to detect with sediment cups.

Cesium-137 measurements in the laboratory require soil sampling in the field. Since alpine sites are heterogeneous (micro morphology) either the number of soil samples per site must be increased or in-situ measurements which smooth out irregularities must be done in order to achieve a representative Cesium-137 activity of a site. The quantification of soil erosion with in-situ Cesium-137 measurements turned out to be a suitable tool for alpine regions. Measurements can be accomplished within a few days to determine long term erosion rates since the Chernobyl accident (deposition of Cesium-137).

7.3 Erosion prediction based on USLE and WEPP

Erosion rates that are calculated with the Universal Soil Loss Equation are in the same order of magnitude compared to Cs-137 based erosion rates on the land use type pasture with dwarf shrubs (paw). On the land use types hayfields and pasture without dwarf shrubs, USLE based erosion values are considerably lower compared to the measured values. The high differences in erosion values are most likely attributed to winter processes. The latter was concluded from great visible damages in early spring just after snow melt. Thus, we suggest an alpine USLE including an alpine factor *W* that allows the user to implement a factor for slopes that are influenced by avalanches and snow gliding processes during winter time. This value can only be considered as a first assumption of an adjusted USLE for alpine environments. More data is needed to validate such an alpine factor. However, this is a first step to solve the problem of erosion modeling in alpine environments.
Because of the uncertainties of simulated winter processes, including snow height and development, temporal snowmelt and water amount distribution, we conclude that WEPP is not a useful tool for alpine regions where winter processes seem to have a great influence on the water balance and erosion processes. WEPP underestimated erosion rates for the long time period from 1986 till 2007 by a factor of 10 to 100. Though erosion rates of the vegetation period turned out to account little to the entire erosion rates of the whole year, the model was able to simulate the erosion rates during the vegetation period in a comparable order of magnitude. Simulated soil moisture and overland flow during the vegetation period was simulated well as well. Winter processes have to be investigated as it is not yet clear what factors of winter processes are triggering those high erosion rates and implemented to the WEPP model in order to provide simulation of erosion under changing land use and climate conditions. Furthermore, it has to be keept in mind that the parameter need for the present form of the WEPP model is high (74 parameters) to run this model successfully. The implementation of triggering erosion processes during winter time will surely enlarge this parameter request. Generally, the comparison of WEPP simulations with our measurements (sediment traps as well as Cs-137) improved the understanding of Alpine erosion processes. Winter processes seems to be important drivers of alpine erosion. Thus, WEPP might be a useful tool to differentiate between confounding factors of erosion in alpine systems.

7.4 Outlook

It could be shown that not all measurement types to detect soil erosion rates in lowlands and low mountain regions are a suitable tool for alpine regions. Winter processes affect installations like sediment traps. This results in a loss of data or in a measurement error due to defect measurement equipment. Cs-137 measurements turned out to be a useful tool to determine soil erosion in steep alpine regions. Still several tasks remain to be pursued:

1. The Cs-137 method needs to be applied for the detections of short term soil erosion. A next step of our work will be to test, if it is possible to measure annual erosion rates with this Cs-137 as well. After the first measurement (that integrates erosion since Cesium-137 release) each further measurement might be subtracted from previous measurements to obtain

temporary erosion information. The prerequisite is to measure at an exact source position for further measurements. The challenge of this further measurement method will be to decrease the measurement error to make sure that further annual measured erosion processes are not within the measurement error.

2. Additional measurements are needed to add an alpine Factor W since empirical equations like the USLE need a substantial data inventory. To enlarge the data inventory future research should aim to include several alpine valleys. This future work should include the investigation of known areas that are affected by snow gliding processes during winter time or by avalanches. This distinction should be done since our first measurements seem to be greatly influenced by avalanches on plot hf1 and hf3. Furthermore, the influence of snow processes and land use types on soil erosion rates have to be pointed out. Thus, it should be clarified if winter processes mask land use type. The knowledge is of great importance for the development of soil conservation strategies.

3. The Influence of snow processes on soil erosion has to be monitored in detail. Physical processes have to be identified. The idea is that one or two hill slopes has to be prepared with temperature sensors in different soil depth, pillows to measure the pressure of snow on the soil, as well as a camera to monitor snow movements with high temporal resolution. The aim of this investigation should be that the influence of soil degradation by freezing and thawing can be estimated. Furthermore, is should be identified how injuries occur. Is it based on the pressure of snow on the soil or is it based on the occurrence on small ice crystals that developed through thawing and freezing. If such small crystals at the intersection snow-soil do exist this might cause the effect of abrasive paper.

4. Based on snow process measurements those processes have to be physically described to extend the winter and snow module of the WEPP model.
The proposed work for the future would help to enlarge the knowledge of soil erosion processes in alpine regions. Furthermore, adjusting both models to alpine regions would advance the possibility to estimate soil erosion rates for the future under changing land use conditions as well as different scenarios of the influence of climate change. Since the two models types are different the possibility is given that estimations based on variable data inventories can be estimated for the future. For a detailed estimation and a good data inventory, the WEPP model would be preferred, whereas for a rough estimation on large sites with low data inventory estimations on soil erosion can by done with the USLE.

REFERENCES

Adams, J.B. et al., 1995. Classification of Multispectral Images Based on Fractions of Endmembers - Application to Land-Cover Change in the Brazilian Amazon. Remote Sensing of Environment, 52(2): 137-154.

Agnesod, G., Lazzaron, R., Operti, C. and Zappa, C., 2001. Accuracy of field spectrometry in estimating Cs-137 contamination in high altitude alpine soils. Radiation Protection Dosimetry, 97(4): 329-332.

Albers, B.P., Rackwitz, R., Schimmack, W. and Bunzl, K., 1998. Transect survey of radiocesium in soils and plants of two Alpine pastures. The Science of the total environment, 216: 159-172.

Alewell, C., Meusburger, K., Schaub, M. and Konz, N., 2007. Effects of land use and climate change on soil degradation: First Results of the case study in the Urseren Valley (Kanton Uri, Switzerland). Forest, Snow and Landscape.

AlpineConvention, 2005. The Multi-Annual Work Programme of the Alpine Conference. Permanent Sectretariat of the Alpine Convention.

Ambuehl, E., 1961. 100 Jahre Einschneien und ausapern in Andermatt. Sonderausdruck aus dem Quartalsheft 4 "Die Alpen".

Appenzeller, C., Beyert, M., Zenklusen, E. and Scherrer, S.C., 2008. Monitoring climate at Jungfraujoch in the high Swiss Alpine region. Science of the total Environment, 391: 262-268.

Arapis, G.D. and Karandinos, M.G., 2004. Migration of Cs-137 in the soil of sloping semi-natural ecosystems in Northern Greece. Journal of Environmental Radioactivity, 77(2): 133-142.

Asner, G.P. and Lobell, D.B., 2000. A biogeophysical approach for automated SWIR unmixing of soils and vegetation. Remote Sensing of Environment, 74(1): 99-112.

Auerswald, K., Kainz, M. and Fiener, P., 2003. Soil erosion potential of organic versus conventional farming evaluated by USLE modelling of cropping statistics for agricultural districts in Bavaria. Soil Use and Management, 19(4): 305-311.

BAFU, 2007. http://www.hydrodaten.admin.ch/d//2087.htm

Bagarello, V., Di Piazza, G.V., Ferro, V. and Giordano, G., 2008. Predicting unit plot soil loss in Sicily, south Italy. Hydrological Processes, 22(5): 586-595.

Baginska, B., Milne-Home, W. and Cornish, P.S., 2003. Modelling nutrient transport in Currency Creek, NSW with AnnAGNPS and PEST. Environmental Modelling & Software, 18(8-9): 801-808.

Bätzing, W., 1997. Kleines Alpen-Lexikon. Beck Verlag.

Beck, H.L. and Gogolak, C., 1972. In situ Ge(Li) and NaI(Tl) gamma-ray spectrometry. In eds H. a. S. Laboratory & U.S.A.E. Comission: pp. 75.

Beven, K.J. and Kirkby, M.J., 1979. A physically based, variable contributing area model of basin hydrology. Bulletin of Hydrologic Sciences, 24: 43-69.

BFS, 2005. Arealstatistik Schweiz: Zahlen-Fakten-Analysen. Bundesamt für Statistik, Neuchâtel.

Boardman, J.W. and Kruse, F.A., 1994. Automated Spectral Analysis - a Geological Example Using Aviris Data, North Grapevine Mountains, Nevada. Proceedings of the Tenth Thematic Conference on Geologic Remote Sensing - Exploration, Environment, and Engineering, Vol I: I407-I418.

Bonnett, P.J.P., 1990a. A review of the erosional behavior of radionuclides in selected drainage basins. Journal of Environmental Radioactivity, 11: 251-266.

Bonnett, P.J.P., 1990b. A review of the erosional behavior of radionuclides in selected drainage basins. Journal of Environmental Radioactivity, 11(3): 251-266.

Bonta, J.V., 2002. Modification and performance of the Coshocton wheel with the modified drop-box weir. Journal of Soil and Water Conservation, 57(6): 364-373.

Boubakari, M. and Morgan, R.P.C., 1999. Contour grass strips for soil erosion control on steep lands: a laboratory evaluation. Soil Use and Management, 15(1): 21-26.

Brazier, R.E., Beven, K.J., Freer, J. and Rowan, J.S., 2000. Equifinality and uncertainty in physically based soil erosion models: Application of the glue methodology to WEPP-the water erosion prediction project-for sites in the UK and USA. Earth Surface Processes and Landforms, 25(8): 825-845.

Clark, M.J. and Smith, F.B., 1988. Wet and Dry Deposition of Chernobyl Releases. Nature, 332(6161): 245-249.

ClimChAlps, 2006. Climate Change, Impacts and Adaption Strategies in the Alpine Space. The ClimChAlp Partnership: 1-32.

Collins, A.L., Walling, D.E., Sichingabula, H.M. and Leeks, G.J.L., 2001. Using C-137 measurements to quantify soil erosion and redistribution rates for areas under different land use in the Upper Kaleya River basin, southern Zambia. Geoderma, 104(3-4): 299-323.

De Asis, A.M. and Omasa, K., 2007. Estimation of vegetation parameter for modeling soil erosion using linear Spectral Mixture Analysis of Landsat ETM data. Isprs Journal of Photogrammetry and Remote Sensing, 62(4): 309-324.

De Asis, A.M., Omasa, K., Oki, K. and Shimizu, Y., 2008. Accuracy and applicability of linear spectral unmixing in delineating potential erosion areas in tropical watersheds. International Journal of Remote Sensing, 29(14): 4151-4171.

De Jong, S.M., 1994. Derivation of Vegetative Variables from a Landsat TM Image for Erosion Modelling. Earth Surface Processes and Landforms, 19(2): 165-178.

De Jong, S.M. et al., 1999. Regional assessment of soil erosion using the distributed model SEMMED and remotely sensed data. Catena, 37(3-4): 291-308.

DecagonDevices, 2007. http://www.decagon.com/ag_research/home/index.php.

Descroix, L. and Mathys, N., 2003. Processes, spatio-temporal factors and measurements of current erosion in the French Southern Alps: A review. Earth Surface Processes and Landforms, 28(9): 993-1011.

Descroix, L., Viramontes, D., Vauclin, M., Barrios, J.L.G. and Esteves, M., 2001. Influence of soil surface features and vegetation on runoff and erosion in the Western Sierra Madre (Durango, Northwest Mexico). Catena, 43(2): 115-135.

Edwards, D.P., Halvorson, C.M. and Gille, J.C., 1999. Radiative transfer modeling for the EOS Terra satellite Measurement of Pollution in the Troposphere (MOPITT) instrument. Journal of Geophysical Research-Atmospheres, 104(D14): 16755-16775.

Elwell, H.A. and Stocking, M.A., 1976. Vegetal Cover to Estimate Soil Erosion Hazard in Rhodesia. Geoderma, 15(1): 61-70.

EnvironmentalProducts, 2006. Analytic of environmental Products, Cottbus. http://www.upgmbh.com/.

Fecht, M., Höfle, B., Starnberger, R. and Kaser, G., 2005. Eine Karte der aktuellen Verdunstung für das Tirol Atlas Gebiet anhand von Landnutzungs- und Vegetationsdaten. Institut für Geographie, University of Innsbruck.

http://tirolatlas.uibk.ac.at/topics/water/pub/evaporation.pdf

Fehr, W., 1926. Geologische Karte der Urserenzone, Beiträge zur geologischen Karte der Schweiz. Kommisionsverlag: A. Franke A.G., Bern.

Felix, R. and Johannes, B., 1995. Bodenerosionsuntersuchungen auf Testparzellen im Kalkhochgebirge. Mitteilungen der Oesterreichischen Geographischen Gesellschaft, 137: 76-92.

Fell, R. et al., 2008. Guidelines for landslide susceptibility, hazard and risk zoning for land-use planning Commentary. Engineering Geology, 102(3-4): 99-111.

Flanagan, D.C. and Livingston, S.J., 1995. WEPP User Summary. Agricultural Research Service, 11: 1-141.

Folly, A., Bronsveld, M.C. and Clavaux, M., 1996. A knowledge-based approach for C-factor mapping in Spain using Landsat TM and GIS. International Journal of Remote Sensing, 17(12): 2401-2415.

forestandrange, 2005. Rangeland Monitoring in Western Uplands.

Foster, G.R. and Highfill, R.E., 1983. Effect of Terraces on Soil Loss - Usle P-Factor Values for Terraces. Journal of Soil and Water Conservation, 38(1): 48-51.

Francis, A.J., 1986. The Ecological Effects of Acid Deposition .2. Acid-Rain Effects on Soil and Aquatic Microbial Processes. Experientia, 42(5): 455-465.

Frankenberg, P., Geier, B., Prostwitz, E., Schütz, J. and Seeling, S., 1995. Investigation on soil erosion and solid matter transport in the Gunzrieder valley/Oberallgäu. Forstwissenschaftliches Centralblatt, 114(4): 213-231.

Fuhrer, J. et al., 2006. Climate risks and their impact on agriculture and forests in Switzerland. Climatic Change, 79(1-2): 79-102.

Gabriels, D., Ghekiere, G., Schiettecatte, W. and Rottiers, I., 2003. Assessment of USLE cover-management C-factors for 40 crop rotation systems on arable farms in the Kemmelbeek watershed, Belgium. Soil and Tillage Research, 74(1): 47-53.

Glenn, N.F. et al., 2005. Hyperspectral data processing for repeat detection of small infestations of leafy spurge. Remote Sensing of Environment, 95(3): 399-412.

Green, A.A., Berman, M., Switzer, P. and Craig, M.D., 1988. A Transformation for Ordering Multispectral Data in Terms of Image Quality with Implications for Noise Removal. Ieee Transactions on Geoscience and Remote Sensing, 26(1): 65-74.

Gronsten, H.A. and Lundekvam, H., 2006. Prediction of surface runoff and soil loss in southeastern Norway using the WEPP Hillslope model. Soil and Tillage Research, 85(1-2): 186-199.

HADES, 2007. Swiss Federal Office for the Environment. University of Bern. http://www.hades.unibe.ch/.

Haigh, M.J., 1977. The use of erosion oins in the study of slope evolution. British Geomorphological Research Group: 31-49.

Hancock, G.R., Loughran, R.J., Evans, K.G. and Balog, R.M., 2008. Estimation of soil erosion using field and modelling approaches in an undisturbed arnhem land catchment, Northern Territory, Australia. Geographical Research, 46(3): 333-349.

Harris, D., Horwath, W.R. and van Kessel, C., 2001. Acid fumigation of soils to remove carbonates prior to total organic carbon or carbon-13 isotopic analysis. Soil Science Society of America Journal, 65(6): 1853-1856.

Harsanyi, J.C. and Chang, C.I., 1994. Hyperspectral Image Classification and Dimensionality Reduction - an Orthogonal Subspace Projection Approach. Ieee Transactions on Geoscience and Remote Sensing, 32(4): 779-785.

Haugen, L.E., 1992. Small-Scale Variation in Deposition of Radiocesium from the Chernobyl Fallout on Cultivated Grasslands in Norway. Analyst, 117(3): 465-468.

He, Q. and Walling, D.E., 2000. Calibration of a field-portable gamma detector to obtain in situ measurements of the Cs-137 inventories of cultivated soils and floodplain sediments. Applied Radiation and Isotopes, 52(4): 865-872.

Heckrath, G. et al., 2005. Tillage erosion and its effect on soil properties and crop yield in Denmark. Journal of Environmental Quality, 34(1): 312-324.

Helming, K., Auzet, A.V. and Favis-Mortlock, D., 2005. Soil erosion patterns: evolution, spatio-temporal dynamics and connectivity. Earth Surface Processes and Landforms, 30(2): 131-132.

Hill, R.A. and Foody, G.M., 1994. Separability of Tropical Rain-Forest Types in the Tambopata-Candamo Reserved Zone, Peru. International Journal of Remote Sensing, 15(13): 2687-2693.

Hofmann, W., Gastberger, M. and Türk, R., 1995. Bodenflechten der alpinen Höhenstufe als Bioindikator für radioaktiven Fallout. Umweltbundesamt.

Huang, C.H., Bradford, J.M. and Laflen, J.M., 1996. Evaluation of the detachment-transport coupling concept in the WEEP rill erosion equation. Soil Science Society of America Journal, 60(3): 734-739.

Isselin-Nondedeu, F. and Bedecarrats, A., 2007. Influence of alpine plants growing on steep slopes on sediment trapping and transport by runoff. Catena, 71: 330-339.

IUSS Working Group WRB, 2006.

Joint Research Center Ispra, 2008. Land Management & Natural hazards unit, Comiss European Communities,. http://eurosois.jrc.ec.europa.eu/library/themes/LandSlides, Acess data: 7.1.09.

Kaegi, H.U., 1973. Die traditionelle Kulturlandschaft im Urserental. Beitrag zur alpinen Kulturgeographie: 212.

Kaste, J.M., Heimsath, A.M. and Hohmann, M., 2006. Quantifying sediment transport across an undisturbed prairie landscape using cesium-137 and high resolution topography. Geomorphology, 76(3-4): 430-440.

Kirkby, M.J., Irvine, B.J., Jones, R.J.A., Govers, G. and Team, P., 2008. The PESERA coarse scale erosion model for Europe. I. - Model rationale and implementation. European Journal of Soil Science, 59(6): 1293-1306.

Kirkby, M.J., Le Bissonais, Y.L., Coulthard, T.J., Daroussin, J. and McMahon, M.D., 2000. The development of land quality indicators for soil degradation by water erosion. Agriculture Ecosystems & Environment, 81(2): 125-136.

Konz, M., Uhlenbrook, S., Braun, L., Shrestha, A. and Demuth, S., 2007. Implementation of a process-based catchment model in a poorly gauged, highly glacierized Himalayan headwater. Hydrology and Earth System Sciences, 11(4): 1323-1339.

Konz, N., Baenninger, D., Nearing, M.A. and Alewell, C., 2009a. Does WEPP meet the specificity of soil erosion in steep mountain regions? Hydrology and Earth System Sciences Discussion, 6(2): 2153-2188.

Konz, N., Schaub, B., Prasuhn, V. and Alewell, C., submitted. Cs-137 based erosion rate determination of a steep mountainous region, Journal of Plant nutrition and soil science.

Konz, N., Schaub, M., Prasuhn, V. and Alewell, C., 2009b. Cs-137 based erosion rate determination of a steep mountainous region. Journal of Plant Nutrition and Soil Science - Zeitschrift fuer Pflanzenernahrung und Bodenkunde accepted, April 2009.

Krause, K., 2003. Radiance Conversion of QuickBird data. Digital Globe. Colorado. USA.

Labhart, T.P., 1999. Planbeilage: Geologisch-tektonische Uebersichtskarte Aarmassiv, Gotthardmassiv und TAvetscher Zwischenmassiv. In: S. Löw and P. Wyss (Editors). Vorerkundung und Prognose der Basistunnels am Gotthard und am Lötschberg. Balkema a.a., Rotterdam.

Laedermann, J.P., Byrde, F. and Murith, C., 1998. In-situ gamma-ray spectrometry: the influence of topography on the accuracy of activity determination. Journal of Environmental Radioactivity, 38(1): 1-16.

Laflen, J.M., Elliot, W.J., Flanagan, D.C., Meyer, C.R. and Nearing, M.A., 1997. WEPP-predicting water erosion using a process-based model. Journal of Soil and Water Conservation, 52(2): 96-102.

Laflen, J.M., Flanagan, D.C. and Engel, B.A., 2004. Soil erosion and sediment yield prediction accuracy using WEPP. Journal of the American Water Resources Association, 40(2): 289-297.

Laflen, J.M., Lane, L.J. and Foster, G.R., 1991. Wepp - A New Generation Of Erosion Prediction Technology. Journal Of Soil And Water Conservation, 46(1): 34-38.

Laternser, M. and Schneebeli, M., 2003. Long-term snow climate trends of the Swiss Alps (1931-99). International Journal of Climatology, 23(7): 733-750.

Ledermann, T. et al., 2008. Erosion Damage Mapping: Assessing Current Soil Erosion Damages in Switzerland. Advances in GeoEcology, 39: 263-283.

Leifeld, J., Zimmermann, M., Fuhrer, J. and Conen, F., 2008. Storage and turnover of carbon in grassland soils along an elevation gradient in the Swiss Alps. Global Change Biology.

Leser, H., Meier-Zielinski, S., Prasuhn, V. and Seiberth, C., 2002. Soil erosion in catchment areas of Northwestern Switzerland Methodological conclusions from a 25-year research programme. Zeitschrift fuer Geomorphologie, 46(1): 35-60.

Leser, H. and Mosimann, T., 1982. Geoökologische Untersuchungen zu Erosionsausmass, Erosionsgefährdung und Regenerationsfähigkeit alpiner Hänge mit Skipistenplanierungen. Fachbeitrag. zur Schweiz- MaB-Information, 10: 27-34.

Lewis, S.A. et al., 2007. Postfire soil burn severity mapping with hyperspectral image unmixing. Remote Sensing of Environment, 108(4): 467-80.

Liu, J.G., Mason, P., Hilton, F. and Lee, H., 2004. Detection of rapid erosion in SE Spain: A GIS approach based on ERS SAR coherence imagery. Photogrammetric Engineering and Remote Sensing, 70(10): 1179-1185.

Livens, F.R. and Loveland, P.J., 1988. The influence of soil properties on the environmental mobility of caesium in Cumbria. Soil Use and Management, 4(3): 69-75.

Machart, P., Hofmann, W., Turk, R. and Steger, F., 2007. Ecological half-life Of Cs-137 in lichens in an alpine region. Journal of Environmental Radioactivity, 97: 70-75.

Mati, B.M., 1994. Splash Transport of Soil on a Slope under Various Crop Covers. Agricultural Water Management, 26(1-2): 59-66.

Matisoff, G., Bonniwell, E.C. and Whiting, P.J., 2002a. Radionuclides as indicators of sediment transport in agricultural watersheds that drain to Lake Erie. Journal of Environmental Quality, 31(1): 62-72.

Matisoff, G., Bonniwell, E.C. and Whiting, P.J., 2002b. Soil erosion and sediment sources in an Ohio watershed using beryllium-7, cesium-137, and lead-210. Journal of Environmental Quality, 31(1): 54-61.

McGee, E.J., Colgan, P.A., Dawson, D.E., Rafferty, B. and Okeeffe, C., 1992. Effects of Topography on Cesium-137 in Montane Peat Soils and Vegetation. Analyst, 117(3): 461-464.

MeteoSwiss, 2007.

Meusburger, K. and Alewell, C., 2008. Impacts of anthropogenic and environmental factors on the occurence of shallow landslides in an alpine catchment (Urseren Valley, Switzerland). Natural Hazards and Earth System Science, 8: 509-520.

Meusburger, K., Baenninger, D., Konz, N., Schaub, M. and Alewell, C., 2009. Estimating vegetation parameters for soil erosion assessment in an alpine region by means of QuickBird imagery. Journal of Applied Earth Observation and Geoinformation, in prep.

Miller, K.L. and Shebell, P., 1993. In situ gamma-ray spectrometry: A tutorial for environmental radiation scientists. In eds E.M. Laboratory & U.S.D. o. Energy, United States: pp 45.

Morgan, M.D., 1995. Modeling Excess Sulfur Deposition on Wetland Soils Using Stable Sulfur Isotopes. Water Air and Soil Pollution, 79(1-4): 299-307.

Morgan, R.P.C., 1994. World Soil-Erosion and Conservation - Pimental, D. Applied Geography, 14(2): 187-187.

Mundt, J.T. et al., 2005. Discrimination of hoary cress and determination of its detection limits via hyperspectral image processing and accuracy assessment techniques. Remote Sensing of Environment, 96(3-4): 509-517.

Nearing, M.A., 1998. Why soil erosion models over-predict small soil losses and under-predict large soil losses. Catena, 32(1): 15-22.

Nearing, M.A., Deerascough, L. and Laflen, J.M., 1990. Sensitivity analysis of the WEPP hillslope profile erosion model. Transactions of the ASAE, 33(3): 839-849.

Nearing, M.A., Govers, G. and Norton, L.D., 1999. Variability in soil erosion data from replicated plots. Soil Science Society of America Journal, 63(6): 1829-1835.

Niesiobedzka, K., 2000. Mobile forms of radionuclide Cs-137 in sandy soils in northeastern Poland. Polish Journal of Environmental Studies, 9(2): 133-136.

Oppikofer, T., Jaboyedoff, M. and Keusen, H.R., 2008. Collapse at the eastern Eiger flank in the Swiss Alps. Nature Geoscience, 1(8): 531-535.

Owens, P.N., Walling, D.E. and He, Q.P., 1996. The behaviour of bomb-derived caesium-137 fallout in catchment soils. Journal of Environmental Radioactivity, 32(3): 169-191.

Pieri, L. et al., 2007. Using the Water Erosion Prediction Project (WEPP) model to simulate field-observed runoff and erosion in the Apennines mountain range, Italy. Journal of Hydrology, 336(1-2): 84-97.

Prasuhn, V., Liniger, H., Hurni, H. and Friedli, S., 2007. Map of soil erosion risk in Switzerland. Agrarforschung, 14(3): 120-127.

Raclot, D. and Albergel, J., 2006. Runoff and water erosion modelling using WEPP on a Mediterranean cultivated catchment. Physics and Chemistry of the Earth, 31(17): 1038-1047.

Renard, K.G., Foster, G.R., Weesies, G.A., McCool, D.K. and Yoder, D.C., 1997. Predicting soil erosion by water;a guide to conservation planning with the revised universal soil loss equation (RUSLE). U.S. Department of Agriculture, Handbook 703: 404.

Richter, R., 2005a. Atmospheric / topographic correction for satellite imagery. Atcor 2/3 user guide. version 6.1, Wessling, Germany.

Richter, R., 2005b. Atmospheric/Topographic correction for satellite imagery. Actor 2/3 user Guide, version 6.1, Wesseling, Germany.

Riesen, T.K., Zimmermann, S. and Blaser, P., 1999. Spatial distribution of Cs-137 in forest soils of Switzerland. Water Air and Soil Pollution, 114(3-4): 277-285.

Ritchie, J.C. and McCarty, G.W., 2003a. (137)Cesium and soil carbon in a small agricultural watershed. Soil & Tillage Research, 69(1-2): PII S0167-1987(02)00127-7.

Ritchie, J.C. and McCarty, G.W., 2003b. (137)Cesium and soil carbon in a small agricultural watershed. Soil & Tillage Research, 69(1-2): 45-51.

Ritchie, J.C. and McHenry, J.R., 1990. Application of radioactive fallout cesium-137 for measuring soil-erosion and sediment accumulation rates and patterns - a review. Journal of Environmental Quality, 19(2): 215-233.

Roberts, D.A., Smith, M.O. and Adams, J.B., 1993. Green Vegetation, Nonphotosynthetic Vegetation, and Soils in Aviris Data. Remote Sensing of Environment, 44(2-3): 255-269.

Robichaud, P.R. and Brown, R.E., 2002. Silt Fences: An Economical Technique for Measuring Hillslope Soil Erosion. United States Department of Agriculture: 1-24.

Rochester, E.W., Hill, D.T. and Yoo, K.H., 1994. Impact of Reservoir Tillage on Runoff Quality and Quantity. Transactions of the Asae, 37(4): 1183-1186.

Rogler, H. and Schwertmann, U., 1981. Rainfall erosivity and isoerodent map of Bavaria. Zeitschrift für Kulturtechnik und Flurbereinigung, 22: 99-112.

Savabi, M.R., 1993. Modeling Subsurface Drainage and Surface Runoff with Wepp. Journal of Irrigation and Drainage Engineering-Asce, 119(5): 801-813.

Savabi, M.R., Young, R.A., Benoit, G.R., Witte, J.M. and Flanagan, D.C., 1995. USDA-Water Erosion Prediction Project (WEPP)-chapter 3: Winter Hydrology. National Soil Erosion Research Laboratory, West Lafayette, Report No. 10.

Sawhney, B.L., 1972. Selective Sorption and Fixation of Cations by Clay-Minerals - Review. Clays and Clay Minerals, 20(2): 93-95.

Schaub, M., Konz, N., Meusburger, K. and Alewell, C., 2009. A measurement routine to determine Cs-137 activities at steep mountain slopes. Soil Use and Management, in review.

Schimmack, W., Bunzl, K. and Flessa, H., 1994. Short-Term and Long-Term Effects of Plowing on the Vertical-Distribution of Radiocesium in 2 Bavarian Soils. Soil Use and Management, 10(4): 164-168.

Schoorl, J.M., Boix Fayos, C., de Meijer, R.J., van der Graaf, E.R. and Veldkamp, A., 2004. The Cs-137 technique applied to steep Mediterranean slopes (Part I): the effects of lithology, slope morphology and land use. Catena, 57(1): 15-34.

Schreurs, M.A., 2007. The politics of acid rain in Europe. Acid in the Environment: Lessons Learned and Future Prospects: 119-149.

Schuepp, M., 1975. Objective weather forecasts using statistical aids in Alps. Rivista Italiana Di Geofisica E Scienze Affini, 1: 32-36.

Schuller, P. et al., 2004. Use of Cs-137 measurements to estimate changes in soil erosion rates associated with changes in soil management practices on cultivated land. Applied Radiation And Isotopes, 60(5): 759-766.

Seibert, J., 2003. Reliability of model predictions outside calibration conditions. Nordic Hydrology, 34(5): 477-492.

Simonato, T., Bischetti, G.B. and Crosta, G.B., 2002. Evaluating soil erosion with RUSLE and WEPP in an alpine environment (Dorena Valley - Central Alps, Italy). Sustainable Land Management - Environmental Protection, 35: 481-494.

Swisstopo, 2006. Reproduziert mit Bewilligung von swisstopo. (BA071108, Zürich).

Theseira, M.A., Thomas, G., Taylor, J.C., Gemmell, F. and Varjo, J., 2003. Sensitivity of mixture modelling to end-member selection. International Journal of Remote Sensing, 24(7): 1559-1575.

Thiam, A.K., 2003. The causes and spatial pattern of land degradation risk in southern Mauritania using multitemporal AVHRR-NDVI imagery and field data. Land Degradation & Development, 14(1): 133-142.

Tiscarenolopez, M., Lopes, V.L., Stone, J.J. and Lane, L.J., 1993. Sensitivity Analysis of the Wepp Watershed Model for Rangeland Applications .1. Hillslope Processes. Transactions of the Asae, 36(6): 1659-1672.

Tyler, A.N., Davidson, D.A. and Grieve, I.C., 2001. In situ radiometric mapping of soil erosion and field-moist bulk density on cultivated fields. Soil Use and Management, 17(2): 88-96.

UP, 2006.

US Department of Agriculture, S.C.S., 1977a. Procedure for computing sheet and rill erosion on project areas. Technical Release, 51((Rev. 2)).

US Department of Agriculture, S.C.S., 1977b. Procedure for computing sheet and rill erosion on project areas, Soil Conservation Service.

Van Dijk, A., Bruijnzeel, L.A. and Wiegman, S.E., 2003. Measurements of rain splash on bench terraces in a humid tropical steepland environment. Hydrological Processes, 17(3): 513-535.

Van Rompaey, A., Bazzoffi, P., Jones, R.J.A. and Montanarella, L., 2005. Modeling sediment yields in Italian catchments. Geomorphology, 65(1-2): 157-169.

Vrieling, A., 2006. Satellite remote sensing for water erosion assessment: A review. Catena, 65(1): 2-18.

Walling, D.E. and He, Q., 1999. Improved models for estimating soil erosion rates from cesium-137 measurements. Journal of Environmental Quality, 28(2): 611-622.

Walling, D.E., He, Q. and Blake, W., 1999. Use of Be-7 and Cs-137 measurements to document short- and medium-term rates of water-induced soil erosion on agricultural land. Water Resources Research, 35(12): 3865-3874.

Walling, D.E. and Quine, T.A., 1990. Calibration of Caesium-137 measurements to provide quantitative erosion rate data. Land Degradation & Development, 2: 161-175.

Wang, B.L. and Cavers, D.S., 2008. A simplified approach for rockfall ground penetration and impact stress calculations. Landslides, 5(3): 305-310.

Wicherek, S.P. and Bernard, C., 1995. Assessment of soil movements in a watershed from Cs-137 data and conventional measurements (example - the Parisian basin). Catena, 25(1-4): 141-151.

Wischmeier, W.H. and Smith, D.D., 1978a. Predicting Rainfall Erosion Losses - A Guide to Conservation Planning. U.S. Department of Agriculture, Handbook 537.

Wischmeier, W.H. and Smith, D.D., 1978b. Predicting Rainfall Erosion Losses - A Guide to Conservation Planning.

WRB, 2006. World reference base for soil resources 2006. 2nd edition. World Soil Resources Reports No. 103. FAO, Rome.

Wu, R.G. and Tiessen, H., 2002. Effect of land use on soil degradation in alpine grassland soil, China. Soil Science Society of America Journal, 66(5): 1648-1655.

Yelpatyevskiy, P.V. and Arzhanova, V.S., 1988. Effect of Acid-Rain on Properties of Mountain Brown Forest Soils. Soviet Soil Science, 20(1): 105-116.

Yu, B. and Rosewell, C.J., 2001. Evaluation of WEPP for runoff and soil loss prediction at Gunnedah, NSW, Australia. Australian Journal of Soil Research, 39(5): 1131-1145.

Zapata, F., 2003. Field application of the Cs-137 technique in soil erosion and sedimentation studies - Introduction. Soil and Tillage Research, 69(1-2): 1-2.

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