> Soil Erosion in the Alps

Experience gained from case studies (2006–2013)

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> Abstracts

This publication of the FOEN “Environmental studies” series gives an overview of the knowledge gained on soil erosion in the Alps during several case studies between 2006 to 2013. Many Alpine areas experience an increase in soil erosion, which is demonstrated for three sites in the cantons Uri, Valais and Ticino. Potential causes for the increased erosion susceptibility as well as controlling factors in general are analysed and suitable methods for soil erosion assessment in Alpine areas evaluated.

Keywords:
Alps, soil erosion, landslide, climate change, land use change

Die vorliegende Publikation in der BAFU-Reihe «Umwelt-Wissen» präsentiert die Erkenntnisse zur Bodenerosion in den Alpen, die in zahlreichen Einzelstudien von 2006 bis 2013 gewonnen wurden. In vielen Alpinen Gebieten ist eine Zunahme der Bodenerosion zu beobachten. Diese wird exemplarisch an drei Gebieten im Kanton Uri, Wallis und Tessin aufgezeigt. Mögliche Ursachen der Zunahme der Schäden, sowie generelle Steuergrössen der Bodenerosion werden analysiert und Methoden zur Bodenerosionserfassung auf Ihre Eignung in Alpinen Gebieten beurteilt.

Stichwörter:
Alpen, Bodenerosion, Rutschung, Klimawandel, Landnutzungswandel

La présente publication qui paraît dans la série OFEV «Connaissance de l’environnement» établit l’état des connaissances sur l’érosion des sols dans les Alpes résultant de nombreuses études réalisées de 2006 à 2013. Une augmentation de l’érosion est constatée dans de nombreuses régions alpines. Cette augmentation est illustré par l’étude de trois régions dans les cantons d’Uri, du Valais et du Tessin. Les diverses facteurs qui pourraient expliquer le constat de l’augmentation de la fréquence des atteintes, ainsi que les facteurs qui contribuent généralement à l’érosion des sols sont analysés. Des méthodes destinées à l’évaluation et la quantification de l’érosion dans les régions alpines sont présentées et commentées.

Mots-clés:
Alpes, érosion des sols, glissements de terrain, changement climatique, évolution des utilisations des sols

La seguente pubblicazione della serie «Studi sull’Ambiente» dell’UFAM offre una panoramica dei risultati di diversi casi studio, svolti tra il 2006 e il 2013, dedicati alla ricerca dell’erosione del suolo nelle Alpi. La ricerca, condotta in tre diversi siti nei cantoni Uri, Vallese e Ticino, evidenzia un aumento dell’erosione del suolo nelle aree alpine. Nel presente studio, le cause potenziali di questo aumento nonchè i principali parametri di controllo del fenomeno vengono presi in considerazione e analizzati. Sono infine descritti e valutati i diversi metodi di stima dell’erosione del suolo nelle Alpi.

Parole chiave:
Alpi, erosione del suolo, frana, cambiamento climatico, cambiamento dell'uso del suolo

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The important study on soil erodibility and rainfall erosivity was only possible within the collaboration agreement (No. 31576) between the European Union (Institute for Environment and Sustainability) and the Swiss Federal Council (Federal Office for the Environment). Thanks go to the Soil Action for their hospitality and in particular to Panos Panagos for the excellent co-operation.

Especially the assessment of the former land use situation was a challenge. For the Obergomer Valley we were supported by Bruno Anthenien, Norbert Agten and Florian Hallenbarter. In the Urseren Valley we were supported by Martin Schaffner from the History Department and Erika Hiltbrunner from the Plant Ecology group of the University of Basel. Moreover, Meinrad Müller from the Korporation Urseren was very helpful throughout the project. Thanks go also to the farmers of the valleys who agreed to the investigation of the sites and were open for interviews.

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> Foreword

In Switzerland, the erosion of arable land has been the subject of numerous studies aimed at understanding its causes and mechanisms. These have led to the introduction of monitoring and prevention measures and the development of tools for estimating the risks. The Guide to Combating the Erosion of Arable Soil (1991) Ordinance Relating to Impacts on the Soil (OIS)(1998), the online map of the potential risk to agricultural soil in Switzerland (2012) and the module on the protection of agricultural soil (2013) all contribute to forming the legislative and technical framework for protecting arable land against water erosion.

Whereas the erosion of arable land and erosion due to natural dangers (such as avalanches, flooding or rockfalls) have been extensively studied, this is not the case as far as Alpine pastures are concerned. Around ten years ago, aware of this lack of knowledge in Switzerland and in the other Alpine countries, the FOEN decided to support research being conducted by the Institute for Environmental Geosciences at the University of Basel that aims to address these deficiencies.

This scientific publication summarises and presents the results of this research, carried out over the past ten years, together with the conclusions reached at two workshops on the same subject. This overview sets out the state of knowledge and paves the way for future developments with a view to improving understanding, and takes account of the risks that erosion poses to Alpine pastures.

Gérard Poffet
Vice Director
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> Summary

Mountain soils are characterized by an intrinsic fragility and low resilience (i.e. the capability to recover after severe stress), making the process of soil erosion almost irreversible.

Until recently, soil erosion research almost exclusively concentrated on arable land. As a consequence, suitable methods to describe and predict soil erosion in mountain areas with low accessibility, steep topography, and extreme climate are missing. Shallow landslides are an evident soil erosion feature in the Alps, but they are traditionally seen as a natural hazard and their relevance for soil loss and the connected on-site damage is rarely considered.

The second major process responsible for soil loss is soil erosion by water, which is the transport of single soil particles. It is influenced by many factors, which can be summarized as rainfall erosivity, soil erodibility, slope steepness, the protection due to soil cover and vegetation, and – in alpine areas – the erosivity of the snow movement and snowmelt.

In the presented work, we aim at a holistic evaluation of methods of assessing soil loss in alpine areas as well as the possible processes and causes responsible for it.

Switzerland is one of the countries where strong climate change impacts are expected. The temperature increase in the Swiss Alps is clearly above the global average and future projections suggest decreasing return periods of extreme rainfall events. In addition, all over the European Alps, major land use changes are observed. Such land-use and climate changes are expected to affect the susceptibility of mountain soils. In order to react to these changes or even to allow their timely prevention, a better understanding and assessment of alpine soil erosion is needed.

Since 2005, the University of Basel (Environmental Geosciences) has been engaged in filling this knowledge gap. New methods of assessing and mapping soil erosion and soil erosion risk factors in the Alps have been developed. They range from remote sensing, modelling, and isotope analysis (^{13}C, ^{15}N, ^{18}O, ^{137}Cs, $^{239+240}Pu$) to field measurements and soil mapping.

Three Alpine valley sites that differ in land-use history, climatic conditions, and geology have been chosen: the Obergoms (65 km^2), Urseren, (30 km^2) and Bedretto/Piora valleys (90 km^2). All three sites show increasing trends of the maximum of cumulative three-day rain events (relevant for landslide triggering). Regarding land use in the Obergoms and Urseren valleys, productive areas are in continuous use, and land use has often been intensified while less productive areas have been abandoned, leading to natural shrub encroachment and reforestation. In the Bedretto/Piora valleys, land use management has remained almost unchanged for decades.

The mapping of shallow landslide incidence over the last 40 to 50 years showed clear increases in landslide-affected areas of 59, 92, and 43% for the Obergoms, Urseren, and Bedretto/Piora valleys, respectively. Based on the inventory maps, shallow landslide susceptibility maps were constructed with a multivariate logistic regression model using various potential causal factors. For the Urseren Valley geology, slope angle and stream density were the most significant parameters. For the Obergoms, geology loses some relevance while snow-related predictors gain importance (slope angle and avalanche density were the most significant predictors [$p < 0.0001$], followed by slope aspect and geologic formation [$p < 0.001$]). The shift towards snow-related predictors is even more obvious for the Bedretto/Piora valleys, where slope aspect, followed by rainfall erosivity and slope angle, is the most important causal factor.

The models were successful in describing the spatial pattern of the shallow landslide occurrence, but could not explain the increase over time. The methods of mapping landslides and landslide susceptibility are well known and used worldwide. However, it was shown that dynamic shallow landslide causal factors such as climate (changes in rainfall and snow patterns) and land use can cause trends and even change the spatial patterns of landslide susceptibility, resulting in a loss of prediction quality for future events. A multi-temporal susceptibility assessment was proposed that can resolve the problem if sufficient land-use information is available.

The locations with increased landslide susceptibility were very different between the three valley areas. The Bedretto/Piora valleys showed the weakest increasing trend, which might be attributed to the very constant and extensive land-use management over the last centuries. Moreover, the increase occurred in elevated sites that have never been used for grazing of livestock. In the Urseren Valley the landslides strongly increased on a very susceptible geological formation located in the intensively pastured areas close to the valley bottom. In the neighbouring Obergoms Valley, this highly susceptible geological formation is hardly affected by landslides because, in contrast to the Urseren Valley, it is forested and not pastured. Another land-use effect was observed for the Obergoms Valley: the landslide increase and the change in pasture stocking since 1976 were significantly related for the managed grassland areas. However, the strongest increase in shallow landslides over time occurred in elevated grasslands that have never been managed. This increase in elevated unmanaged areas was also observed in the Bedretto/Piora valleys, indicating altered and higher landslide susceptibility, potentially caused by changing snow conditions. This effect could not be observed for the Urseren Valley because the high altitude sites above the tree line are located on a flat slope section.

Concluding from these studies and literature, an increase of shallow landslides is observed in alpine areas. Both land-use intensification and climate effects are expected to enhance landslide susceptibility. On top of this, the higher frequency and intensity of rainfall events and the strong temperature trends, which affect the snow patterns and dynamics, seem to accelerate soil loss in susceptible areas.

Regarding water-induced soil erosion, measurements across the Alps showed, in agreement with our measurements, that an intact vegetation cover almost prevents any soil loss. However, on grassland plots with clear signs of degradation (reduced vegetation cover), very high erosion rates of >20 t ha^{-1} yr^{-1} can be observed. Considering average values at catchment scale, the water erosion rate is as much as twice as high (1.18 t ha^{-1} yr^{-1}) as the soil loss caused by shallow landslides (0.6 ha^{-1} yr^{-1}).

Investigations in the Urseren valley show that snow-induced soil erosion (through wet avalanches and snow gliding), with maximum values of 23.6 t ha^{-1} $winter^{-1}$, is of the same order of magnitude as maximum rates of water soil erosion during the summer season found in the literature. Interestingly, the rate of snow gliding is strongly determined by slope aspect and land use/land cover. For vegetation with lower surface roughness, higher snow-glide rates and higher “winter” soil erosion rates were found. Consequently, a detailed mapping of vegetation types and vegetation cover, as was done in this study by QuickBird imagery, is most effective for soil erosion assessment. The erosivity induced by the mechanical impact of the snow cover needs further investigation and quantification in order to allow a spatial assessment and integration of this process into soil erosion models.

Processes of soil erosion are highly event based. Tracking erosion via monitoring always requires high temporal resolution to capture peak events for long time periods and also to capture events with a small return period. Classical plot measurements require a long period of continuous measurements to track soil degradation. Thus, new approaches for a better qualitative detection and quantification of soil degradation in alpine ecosystems were investigated.

Fallout radionuclides, particularly ^{137}Cs, are used worldwide for soil erosion quantification. Assessment of soil redistribution rates according to this approach is commonly based on a comparison of the radionuclide inventory (areal activity density) at individual points in the landscape with that of a “stable” landscape position (also termed a reference site), where neither erosion nor deposition has occurred, assuming a homogenous fallout. Fallout radionuclides offer a retrospective soil erosion rate because they give a time-averaged estimate of soil movement since fallout commenced in the mid-1950s due to thermonuclear weapon tests (in the case of $^{239+240}Pu$) and later due to the nuclear power plant accidents such as Chernobyl (in the case of ^{137}Cs). Another great advantage of radionuclides compared to classical methods is that soil erosion by all agents, for example, wind, water, snow, and so on, are captured. The applicability of this method strongly depends on the successful identification of an undisturbed reference site. The latter is particularly problematic for ^{137}Cs due to the general heterogeneous distribution of atmospheric ^{137}Cs Chernobyl fallout and the fact that the ground in Alpine areas was partly snow covered during the fallout event in April 1986, which resulted in an inhomogeneous ^{137}Cs distribution. The stable isotope composition of a soil profile can provide valuable information on the disturbance or non-disturbance of a site which can be used if quantification of soil erosion via radionuclides such as ^{137}Cs and $^{239+240}Pu$ is targeted.

Different model applications – the Water Erosion Prediction Project (WEPP), the Pan European Soil Erosion Risk Assessment (PESERA), and the Revised Universal Soil Loss Equation (RUSLE) – showed that soil erosion-risk assessment benefits more from improving spatial information for model input and validation than from enhancing the complexity of the model. For instance the RUSLE application reproduced the spatial pattern and the relative magnitude best for our sites; it could be a good starting point for water soil erosion assessment and, in the future, could be adapted to include the snow erosivity.

Further research should be dedicated to the regionalization of vegetation cover and rainfall erosivity, qualification of erosion rates in alpine areas, with special focus on winter soil erosion rates, and the interaction of soil, plant, and snow, which is a causal factor for both shallow landslides and topsoil removal.

Even though soil loss in steep alpine areas is inevitable, the magnitude can be strongly affected by preservation of the vegetation cover through careful pasture management. Some of the most important parameters ensuring an intact vegetation cover are animal species, animal behaviour, animal live weight, stocking rate and density, pasture system, and intensity of fertilization.

Sustainability of soil use and management in alpine areas can be achieved if rates of soil loss do not exceed soil production rates, which range approximately from 0.5 t ha^{-1} yr^{-1} to a maximum for young soils of 9 t ha^{-1} yr^{-1}. The soil erosion rates found in our site and other parts of the Alps showed that this rate is frequently exceeded and thus sustainability is not ensured.

> Introduction

Soil erosion: a serious disturbance for (sub-)alpine ecosystems?

Alpine soils – an important resource

Alpine soils are an often undervalued or even overlooked resource. Alpine soils that are mainly managed as forests and grasslands cannot compete with the economic output of arable lowland soils. However, the Alpine soils deliver valuable services that reach far beyond into the lowland, such as water supply, water retention (to prevent flooding of river plains), and recreational services. However, economic, societal, and environmental changes are often an immediate threat to mountain systems and thus careful planning is needed. One inherent parameter of ecological stability is the status of soils in the ecosystems. And even though the stability of mountain soils has always been threatened by natural hazards and erosion, the combined effects of climate- and land-use changes are expected to increase future soil erosion in Alpine regions (Frei et al. 2007).

Land-use change

Alpine soils have been managed by humans for about 5000 years (Bätzing 2005). The prevailing land use in this landscape is grassland farming; for example, 27% of Switzerland is grassland, 72% of the agricultural land is grassland, and 46% of grassland is alpine grassland (BFS 2005). Grassland is also the predominant land-use type at altitudes above 1500 m (a.s.l.) (BFS 2005). Thus, grassland management determines soil and slope stability in the Alps to a large extent.

The European Alps have experienced substantial changes in land cover and land use during the last centuries. Great efforts have been made to maintain soil and slope stability for food production. However, particularly in the past 30 years, economic conditions have changed due to the reduction of protective duties and the resulting liberalization of the agricultural market and increase in competition in global agriculture. In such a globalized market, mountain regions are uncompetitive because of higher production costs (Streifenede and Ruffini 2007). Consequently the abandonment of alpine pastures has increased. The Swiss government aims to prevent land abandonment in mountain regions by subsidies, which consist of basic payments for farmers in mountain regions and rewards for best management practice and extensive use (Flury et al. 2005). Nonetheless, the number of farms decreased by 40% between 1985 and 2008 (BFS 2009), while the land area managed by single farms increased steadily (BFS 2009). Easily accessible sites with high productivity experienced intensification while remote areas with high production costs became marginalized (Bätzing 2005).

The land use and management also affect soil erosion and the incidence of shallow landslides (Schauer 1975; Bunza 1989; Glade 2003; Tasser et al. 2003; Petley et al. 2007). Shallow landslides are frequently observed on steep alpine slopes and may constitute a large proportion of total soil loss. Regarding landslide susceptibility, prevailing land-use developments, abandonment, and intensification are all expected to have an increasing effect (Douglas et al. 1996; Andre 1998; Tasser and Tappeiner, 2002; Krohmer and Deil 2003; Meusburger and Alewell 2009). In the case of abandonment, a decreased stability is expected until succession to larger shrubs and forest occurs (Tasser et al. 2003).

Climate change

In addition to land-use and land-cover changes, European mountain systems have been and will be also confronted with climate change. Recent climate change projections indicate that mountains will be the most vulnerable region of Europe (Schroter et al. 2005). Thus, Switzerland is one of the countries where the strongest effects of climate change are expected (Beniston 2006). Besides the temperature increase, increased frequency and intensity of torrential rainfall events are also expected (IPCC 2007). The latter may affect the water balance and runoff characteristics, resulting in altered sediment yield and frequency of landslides (Asselman et al. 2003; Horton et al. 2006; IPCC 2007). For instance, a more than 200% increase in sediment delivery to the river Rhine from Alpine regions is predicted for the year 2100 (Asselman et al. 2003).

Rising air temperature, especially the increasing number of days with air temperatures above zero, also influences the occurrence of snowfall and time of snowmelt (Birsan et al. 2005). Changes in snowmelt are expected to be critical for altitudes between 500–800 in winter and between 1000–1500 m a.s.l. in spring; Wielke et al. 2004). While snow depths decreased in the late 1980s and 1990s at low elevations (<1000–1300 m a.s.l. in January–February; Laternser and Schneebeli 2003; Beniston 2006), an increase was observed at high elevations (>2000 m a.s.l.; Beniston 2006). Snow melting, which may produce significant surface runoff and related soil erosion, is reported to occur earlier in spring due to rising temperatures but with no obvious shift of snow accumulation in autumn (Laternser and Schneebeli 2003). With the projection of further warming, the duration of snow cover is predicted to be shortened by more than 100 days, with earlier snowmelt in spring (Jasper et al. 2004; Beniston 2006; Horton et al. 2006). A permanent snow cover in winter protects the soil from freezing, which is important for snowmelt infiltration and runoff, respectively (Stahli et al. 2001; Bayard et al. 2005). In conclusion, the higher levels of rainfall and changes in freezing–thawing cycles can be expected to increase soil erosion and mass movement because of the sparsity or absence of vegetation cover at low elevations in winter and early spring (Scheurer et al. 2009).

Processes responsible for soil loss in mountain areas

Soil erosion in Alpine grasslands is mostly driven by two processes: (i) shallow landslides, which are a type of mass movement, and (ii) sheet erosion, which is the gradual detachment and transport of single soil grains and aggregates. While shallow landslides are an evident soil-erosion feature, sheet erosion often happens unseen. The higher visibility of landslides compared to sheet erosion might also be the reason why less attention is paid to the latter. Sheet erosion is a well-studied research topic in lowlands; however, only a few studies on sheet erosion in alpine regions exist even though from a quantitative perspective soil loss by sheet erosion predominates landslide impact in many areas. Differentiating between these two processes is crucial for the assessment of deterioration extent or, on the other hand, of possibilities for sustainable management. Moreover, assessments of soil erosion (by water) and shallow landslide susceptibility require different techniques and tools. The different processes of soil loss will be addressed in more detail in the following chapter.

Mountain soil erosion compared to lowlands

Erosion processes in alpine regions differ from lowland soil erosion in many aspects: soils are less developed, they are exposed more intensively to freezing-thawing, snow-cover, and snow melting processes, and they are exposed to extreme climate and topography. Often a high infiltration rate is observed, which yields moderate or low overland flow. Another major point is that, in comparison to cultivated lowlands, soil-erosion damages in mountain ecosystems are a severe problem because once soil erosion has started, it is very often impossible to stop the process again or great effort is required to do so. Moreover, because of the extreme climate conditions, re-vegetation usually takes longer than in lowlands. Thus, assuming a slow rate of soil formation, soil erosion causes irreversible damages on the time scale of 50 to 100 years (Van der Knijff et al. 2000). Last but not least, mitigation options in lowlands might counteract soil degradation, while melioration of soils in alpine environments is hardly ever possible.

Soil erosion assessment is difficult

Today, soil erosion and its associated impacts are among the most important and yet probably least known environmental problems (Guardian 2004), especially for mountain regions. In mountainous regions, soil erosion quantification under natural precipitation regimes is challenging and thus only a few studies exist (Felix and Johannes 1995; Descroix and Mathys 2003; Isselin-Nondedeu and Bedecarrats 2007). Classical techniques to measure soil erosion such as sediment traps require a long period of continuous measurements, impeding estimates for larger regions. Thus, regional soil erosion assessments for the Alps are based on models that have been developed for lowlands and do not consider soil loss by snow processes and landslides. Therefore a serious validation of these models is needed but is hardly feasible due to difficulties related to the measurement of soil erosion. New approaches for better spatially explicit detection and quantification of soil degradation in alpine ecosystems are therefore urgently needed (EEA 2000).

Research needs and first steps

Because of the abovementioned reasons, soil erosion, especially the quantification of soil erosion rates in the Alps, is and will be a relevant issue. Soil erosion in the Alps was also identified as a priority for action by the soil protocol of the Alpine Convention (Alpine Convention 2005), but a comprehensive assessment of soil erosion in the Alps is still missing (ClimChAlps 2006). In fact, very little is known about current quantities of soil erosion and the relation to different soil erosion causal factors is not yet well understood. Even more demanding is the identification of trends since no monitoring tools exist. The main problem here is that rates of water erosion and landslide occurrence may respond in a non-linear manner to triggering factors such as increases in rainfall (Helming et al. 2005). A separation and quantification of the effect of the different driving factors involved is therefore a great future challenge.

To assess landslide-risk areas, most regions have already developed methods and in the framework of the Interreg IIIB CatchRisk project[1] efforts have been made to exchange and standardize methods to improve landslide prediction. An open question of the CatchRisk project was how to judge and quantify the impact of land use. Also within the SilvaProtect project[2] endeavours have been made to create a Swiss vulnerability map for natural hazards including the process of landslides. However, the prediction of

[1] www.alpinespace.org
[2] www.bafu.admin.ch/naturgefahren/01920/01964/index.html?lang=de

shallow landslides posed problems and especially in the Alpine region the correspondence between observed and modelled instabilities was deficient because of the insufficient spatial resolution of input parameters.

Sheet erosion on alpine grassland can be considered marginal as long as an intact vegetation cover prevents soil loss (Merz et al. 2009b; Martin et al. 2010; Wildhaber et al. 2012). However mismanagement especially in steep areas can rapidly change the situation (Sutter, 2007; Sutter & Keller 2009). Soil loss due to unadapted land management is undesirable. Thus, the Ordinance Relating to Impacts on the Soil (German: Verordnung über die Belastung des Bodens VBBo, Art. 6) and the ordinance concerning the summer pasturing contribution (German: Sömmerungsbeitragsverordnung SöBV, Art. 3, 12) aim to prevent management-related soil erosion to ensure a sustainable use of the soil resource. The implementation of this legislation requires that the natural and anthropogenic impacts on soil erosion be separated.

While agricultural sites in Switzerland have been well investigated for decades and recently a high-resolution soil erosion risk map of Switzerland was successfully implemented as a strategic policy support system (Prasuhn et al. 2013), soil erosion research in the Alps can still be considered as a very young field of research. Only a few research groups have focussed on this topic so far and there is still a great need to test and develop methodologies for soil erosion assessment in alpine areas in order to improve the knowledge of processes and related rates of soil loss. This report will provide an overview of the experience gained in several projects related to the topic at the University of Basel since 2005.

Different soil erosion processes require different methods

Processes of soil erosion

Soil erosion is defined as the displacement of soil particles by the agents tillage, wind, water, and snow and by down-slope movement in response to gravity (Ahnert 2003). Tillage erosion is of minor importance for sub-alpine and alpine regions and the role of wind transport is so far unknown. Typical forms of soil erosion by water are sheet, rill, and gully erosion. Soil particles are detached either directly by means of rain splash or indirectly by surface flow. In alpine grasslands, rill and gully erosion is seldom observed (Strunk 2003). Typically rill and gully erosion occurs on ploughed fields or deep developed soils with scarce vegetation in arid or semi-arid regions (Vrieling et al. 2007). Thus, the dominant but often invisible process of water erosion is sheet erosion (Fig. 1 A). A synonymously used term is “inter-rill erosion”. Sheet erosion is largely enhanced by the reduction of vegetation cover; as such, overgrazing and cattle trails (Fig. 1 B) are supposed to have a strong triggering effect. Moreover, snow cover is an important attribute of alpine areas. Large areas are covered with snow for more than half a year. While snow melt is a well known process of soil erosion, transport and detachment of soil material by the down-slope movement of the snow are not in the scope of soil erosion assessment so far. Particularly wet avalanches can yield enormous erosive forces that are responsible for major soil loss (Gardner 1983; Ackroyd 1987; Bell et al. 1990; Jomelli and Bertran 2001; Fuchs and Keiler 2008; Freppaz et al. 2010; Ceaglio et al. 2012). Another important process of snow movement affecting the soil surface is snow gliding (In der Gand and Zupancic 1966). Snow gliding is the slow (millimetres to centimetres per day) downhill motion of a snowpack over the ground surface caused by the stress of its own weight (Parker 2002). Field observation of dirty snow and soil/vegetation rolls that are shaped by the pressure of the moving snow that is rolling up the vegetation and topsoil layer suggests that this might be a major causal factor for soil erosion. But so far, this process has scarcely been measured (Meusburger et al. 2013a) and has not been implemented in soil erosion models (Fig. 1 D).

Fig. 1 > Typical soil erosion features in alpine areas

Shallow transitional landslide (A), sheet erosion (B) enhanced by cattle trails (C), soil erosion after snowmelt and soil rolls with vegetation residuals formed due to the slow movement of snow on the soil surface (D), and soil material eroded and transported by avalanches (E).

Given that some methods are suited to quantify particular processes (e.g. sediment traps for sheet or rill erosion) and others to quantify overall net losses (e.g. inventories of ^{137}Cs), it is likely that the combination of approaches employed will yield additional knowledge about the relative importance of the different soil erosion processes. The advantages and disadvantages of the different methods and their suitability for alpine regions will also be discussed.

The different processes of soil erosion mentioned above require different methods for their assessment. Sheet erosion can be assessed with sediment traps, rainfall simulations, and fallout radionuclides. Visible mapping of sheet erosion, which is successful for arable land (Prasuhn 2003), is not feasible on grasslands, since visible mapping assesses rill and gully erosion, which is not well developed in alpine grasslands.

Sediment traps provide a measurement of the soil erosion rate for a limited period (usually some years) and area. If either the location or the period is not representative, the rates can be misleading. Rainfall simulations can be used to account for a missing extreme rainfall event even though the comparability to natural hydrological settings is limited especially on steep slopes. As an alternative, fallout radionuclides (FRNs) in general, and particularly ^{137}Cs, have been used worldwide to evaluate net soil erosion rates (Zapata 2003; Mabit et al. 2008). ^{137}Cs is an anthropogenic isotope produced by the testing of thermonuclear weapons during the 1950s and 1960s. When ^{137}Cs fallout reaches the soil surface, it is tightly adsorbed by fine soil particles (Tamura 1964; Tamura and Jacobs 1960) and its subsequent redistribution is associated with soil erosion (Ritchie and McHenry 1990). The ^{137}Cs method has the major advantage of integrating different erosion processes over a timespan of several decades (Mabit et al. 2008). Further, soil erosion can be modelled with empirical models like the (Revised) Universal Soil Loss Equation (RUSLE) or with physically based models that simulate the water flow and related particle transport such as the Water Erosion Prediction Project (WEPP). Both types of models yield an estimate of soil erosion rate, but for RUSLE it is the long-term average gross soil erosion rate while for WEPP it is the net erosion rate associated with the climate period simulated. WEPP does take into account soil erosion by snow melt; however soil removal caused by snow movement – either snow glide or avalanches – is not considered. Recently the first physically based attempts to model the erosive force of wet avalanches were made (Confortola et al. 2012). No similar model for snow gliding exists yet. However, the potential maximum snow glide distance during a targeted period can be modelled with the empirical spatial snow glide model (SSGM) (Leitinger et al. 2008). The modelling of these processes is important to evaluate the impact of the snow glide process on soil erosion at larger scale in mountain areas. Snow processes can be assessed in the field by mapping and measuring the sediment load of accumulated snow/soil fans related to the specific catchment area from which the snow originated. Again, radionuclides will capture this process. However, due to their integrative nature the different processes cannot be separated.

Mass movements

Landslides are a major hazard in mountain regions as they cause large damage to infrastructure and property (Metternich 2005). Landslides are a type of mass movement, which is the down-slope movement of earth materials under the influence of gravity (Ahnert 2003). Material is mobilized when the shear stress imposed on a surface exceeds the shear strength. The – often very sudden – movement of soil packages occurs along a failure plane, which may be a layer of clay or rock upon which the destabilized surface material sets. Landslides can be triggered by (natural) physical processes such as heavy or prolonged rainfall, earthquakes, snow melt, and slope toe erosion by rivers and man-made activities such as slope excavation and loading, land-use changes, water leakage, and so on or by any combination of natural and man-made processes (Joint Research Center Ispra 2008). In affected areas, landslides are also a major source of soil erosion (Joint Research Center Ispra 2008). On steep slopes, various types of mass movements occur, ranging from rock slides, debris flows, and deep landslides to small and shallow erosions and landslides. A shallow landslide is one in which the sliding surface is located within the soil mantle or weathered bedrock (typically to a depth of a few decimetres to some metres). These shallow landslides, also known in German as *Blaiken*, are characterized by small spatial dimensions of about 2 to 200 m² and depths of a few decimetres but usually occur in clusters on a single slope (Wiegand and Geitner 2010). Regarding soil loss, shallow landslides in particular play an important role in alpine areas; thus the following investigations focussed on this type of mass movement.

Contrasting methods are used for landslide susceptibility assessment. Landslides are a visible soil erosion feature that can be mapped in the field or by remote sensing techniques. Regarding the latter, aerial photographs have been applied frequently and successfully (Parker 2002; Meusburger and Alewell 2008; Wiegand et al. 2013), but various satellite imagery has also proved useful for landslide mapping (Shrimali et al. 2001; Liu et al. 2004). The resulting landslide inventories are often incomplete and the events that triggered the landslide cannot be assigned. The scarce information on the time span and the dimensions of the landslides make a conversion to soil erosion rates difficult. This and the probabilistic nature of landslides are the main reasons why different models are used for spatial landslide susceptibility assessment. Even though there are physically based models available their performance is questionable and the data demands immense. Thus, statistical models to predict landslide susceptibility are used most frequently (Carrara et al. 1998; Guzzetti et al. 1999; Guzzetti et al. 2006).

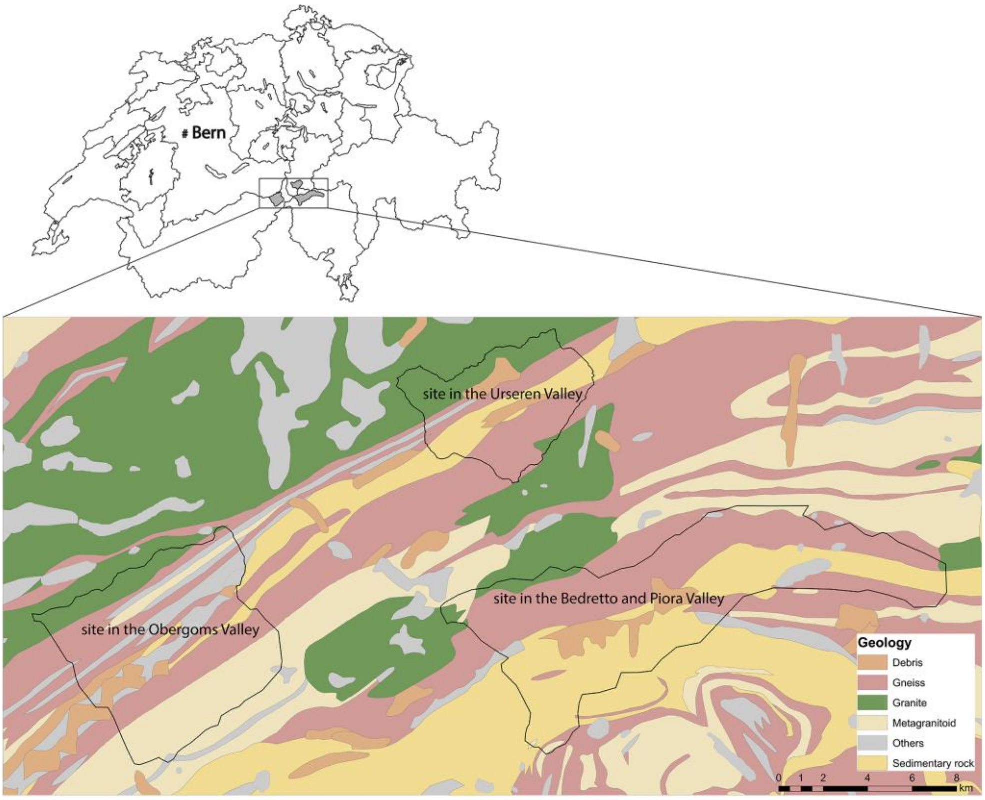
Although both processes – sheet erosion and shallow landslides – are relevant for soil erosion in mountain areas, the intrinsically different nature of the methods and models needed impedes a combined assessment. Consequently when comparing different erosion rates it is important to keep in mind that different processes on different spatial and temporal scales are involved.

1 > Soil erosion in three Alpine valley areas: Evaluation of the current status

Site description

The occurrence of soil erosion depends strongly on catchment characteristics like geology, topography, and so on. However, causal factors that are variable in time such as climate and land use may also have an impact on the recent and future landslide susceptibility and soil erosion risk. To identify the relative importance of land use and climate change as compared to inherent characteristics as causal factors, sites in three Alpine valley areas that differ in land use and climatic conditions are compared (Fig. 2): Obergoms (66 km²), Urseren (34 km²), and Bedretto/Piora (93 km²).

Fig. 2 > Location and geology of the three investigated valley areas



The Obergoms Valley

The sub-alpine study area located in the Obergoms Valley (Canton Valais, north Central Alps; Fig. 2) is discharged by the River Rotten (Rhone). The wide glaciated valley is characterized by a U-shaped profile and a rugged terrain. Elevation ranges from 1334 to 3084 m a.s.l. (mean elevation 1993 m). The predominant slope angle is approximately 25.7° (ranging from 0 to 67.8°). Average annual temperature at the valley bottom is 3.8 °C (Ulrichen 1981–2009). Average annual precipitation varies between the valley bottom (1200 mm) and mountain ridges (1900 mm). Soil types in the catchment are dominated by Podsols, while Leptosols can be found at mountain ridges and slopes and Cambisols and Fluvisols at the valley bottom. The geology of the valley is formed by Aare granite in the north and gneiss and mica schist in the south. On the south-facing slopes, instable layers formed by sedimentary layers from the Mesozoic and Permian are prone to erosion. Large parts of the valley slopes are forested to protect them from avalanches. A large percentage of the valley bottom is under arable use.

The Urseren Valley

The Urseren Valley is located eastwards of the Obergoms valley in Central Switzerland (Canton Uri, Fig. 2). Altitude ranges from 1440 to 3200 m a.s.l. At the valley bottom (1442 m a.s.l.), average annual air temperature for the years 1980 to 2012 is around 4.1 ±0.7 °C and the mean annual rainfall precipitation reaches about 1457 ±290 mm, with 30% falling as snow (MeteoSwiss 2013). The U-formed valley is snow-covered from November to April. Mean annual snowfall (as the sum of precipitation from December to April) is roughly 30% of the annual precipitation (based on the data of MeteoSwiss). The predominant soils are Cambisols and Podzols (anthric) based on the IUSS Working Group (2006) classification. Most of the soils are characterized by a migration horizon (M) which has a typical thickness of 5 to 45 cm and have soil textures of sandy loam or loamy sands. The valley was almost completely deforested in the eleventh century by the Romans and has probably been prone to erosion ever since. In the last decades anthropogenic activity increased on the lower slopes and decreased in the higher, more remote areas (Meusburger and Alewell 2008). The vegetation cover consists of pastured grasslands that are, to a large extent, mixed or even dominated by dwarf shrubs. The proportion of forests (which protect the slope from avalanches) represents only 1% of the surface. Because of the intensive deforestation of the valley, the frequency of avalanches is relatively high (Meusburger and Alewell 2008). The geological setting is comparable to the Obergoms Valley; however in the latter the fragile Mesozoic layer is forested while in the Urseren Valley it is intensively pastured.

The Bedretto/Piora valleys

The third study site is located at the southern part of the Alps (Canton Ticino, south Central Alps; Fig. 2). Two valleys are considered in this region: the Bedretto and Piora valleys. Piora valley (22.6 km²) is a high valley where the elevation ranges from 1850 to 2773 m a.s.l. The average annual precipitation is between 1500 and 1750 mm with approximately 35% falling as snow (based on data of MeteoSwiss 2013). The bedrock is dominated by mica schist and gneiss with small sediment layers and areas of granites (Gotthard Massif in the north, Lukmanier Massif in the south). Similar to the Urseren and Obergoms valleys, in between the two massifs, sedimentary material of the "Piora-Mulde", which became famous in the context of the Gotthard-Tunnel, constitutes the valley floor. Soils of the catchment are mainly Podzols and dystric Cambisols or cumulic Anthrosols with a soil texture of mainly sandy loam to loam. Streets and paths mostly located at the bottom of the south-facing slopes are often prone to avalanches (Knoll-Heitz 1991). This valley was also deforested by the Romans and pasturing has been the dominant land use ever since. However, in contrast to the other valleys, land-use change plays a minor role since the management has been very constant over the centuries due to regulation guidelines, established in the year 1227, regulating the alp zoning and stocking and due to the fact that the valley was managed only by a single alp farm (Knoll-Heitz 1991).

The Bedretto Valley (75km²) in northern Ticino extends from Airolo at the foot of the Gotthard Pass (which crosses into the Urseren Valley) up to the Nufenen Pass, where it crosses into Valais. The elevation ranges from 1400 m to the 3190 m of the Rotondo peak.

All the investigated valleys are characterized by similar geology, with granite gneisses in the summit areas, schists and gneisses along the valley shoulders, and Triassic metamorphic sediments at the valley floor, except for the southern side of the Bedretto Valley, which consists of Penninic sediment and basement (Ustaszewski et al. 2008). The predominant schistosity in all valleys is vertical and parallel to the valley axes (Ustaszewski et al. 2008). For hundreds of years, there was only extensive cattle farming due to a low population density in the valley. The alpine pastures of the Leventina were already divided among the communities in 1227. Large avalanches are frequent in the valley, even though 21% of the area is forested. Approximately 50% of the valley consists of grassland and 6% of shrubs, while a large proportion (20%) is unproductive because it consists of rocks and debris.

1.1 Observations of an increase in shallow landslide incidence

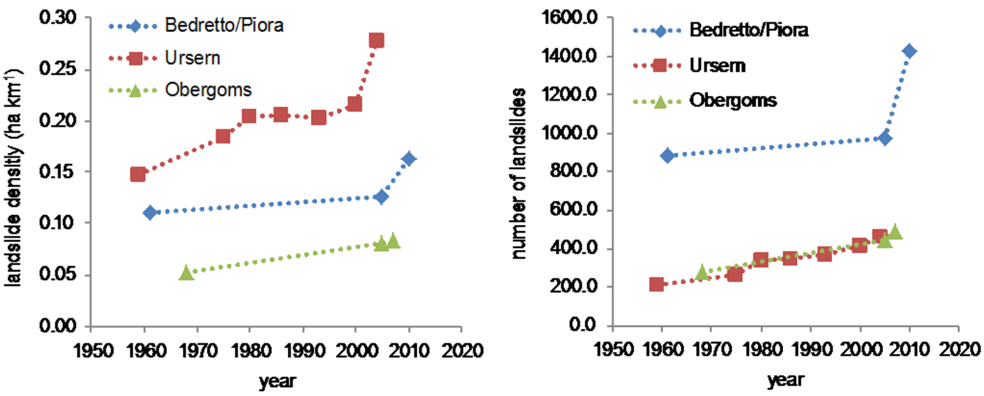
Development of landslide incidence over the last decades

For these valleys the development of shallow landslide susceptibly over the last five decades was mapped by aerial photograph interpretation. The first series of aerial photographs for the Obergoms Valley was available in 1967/1968. For this year, 221 landslides were mapped. The next inventory is based on orthophotos taken in 2005 and 2007 (note that only one other series of photos was available for the year 1987 after the torrential event on 24/25 August). These series could not be used for the landslide inventory mapping since it was not possible to separate large depositional areas from erosion areas. The number of landslides increased to 446 in 2005 and further to 487 in 2007. The latter constitutes a total eroded area of 5.58 ha. An increase of 59% in the eroded area occurred within the 39 years between the first and last landslide inventory maps. The average size (120 m²) of the shallow landslides hardly changed over time.

The greatest increase in shallow landslide incidence was observed for the Urseren Valley. In 2004 a total of 383 shallow landslides (>25 m²) with a total area of 9.42 ha, an average size of 250 m², and an average slope of the landslide area of 33.9° were mapped. Due to the existence of aerial photographs for seven years (1959, 1975, 1980, 1986, 1993, 2000 and 2004) the Neumann-trend test could be applied and showed significant increasing trends ($P < 0.01$) in landslide number and area. While the number of landslides continuously increased with time, the increase in the eroded area happened in two phases: from 1959 to 1980 and from 2000 to 2004. For the period 1980 to 2000 only a small change in the landslide inventory maps was observed. From 1980 to 2000, some new landslides occurred but the total affected area did not increase due to partial regeneration of older landslides. In total, the eroded area nearly doubled between 1959 and 2004 (increase of 92%). The existence of several photos for this region allowed the mapping of different stages of landslide incidence and revealed that, once the slope has been degraded by trails and landslides, it might take decades to recolonize the bare spots with vegetation.

The Piora/Bedretto valleys are characterized by a medium landslide density (0.16 ha/km² compared to 0.08 and 0.28 ha/km² for the Obergoms and Urseren valleys, respectively). Only a slight increase in landslide density was observed between the years 1959/1961 and 2005, but a very pronounced one was observed between 2005 and 2010. In total the area eroded by landslides increased by 43% in the 49 years that are covered by the aerial photographs. Similarly to the Obergoms Valley, the average landslide area is rather small, but while for 2005 the average size was still 120 m² it decreased to 106 m² in 2010, indicating that the most recent landslides are smaller. The Obergoms and Urseren valleys show a very similar gradient in the increase of landslide numbers. In the Urseren Valley the landslide density suggests that the increase occurred in two phases, for example due to two torrential events. However, the continuous increase in the number of landslides points towards a gradual change in landslide susceptibility as is expected from land-use change. In the Bedretto/Piora valleys the dynamic was different. From the limited data available it seems that the period before 2005 was more stable, as in the other valleys, followed by very erosive conditions in the most recent years.

Fig. 3 > Development of shallow landslide density (left) and number (right) for the sites located in the Obergoms, Urseren, and Bedretto/Piora valleys



The observed increase in shallow landslide occurrence is congruent with other studies; for example, Wiegand (2010) reviewed 13 studies on the development of shallow landslides in alpine regions with the result that in recent decades there has been an increased occurrence of shallow erosion on grasslands and pastures of the Alps. In other mountain ranges similar processes are observed, for example in the Apuan Alps (Giannecchini 2005), Pyrenees (GarciaRuiz et al. 1996; Begueria 2006), Japan (Marui 1984; Hiura 1988), and New Zealand (Derose et al. 1995; Brooks et al. 2002; Glade 2003).

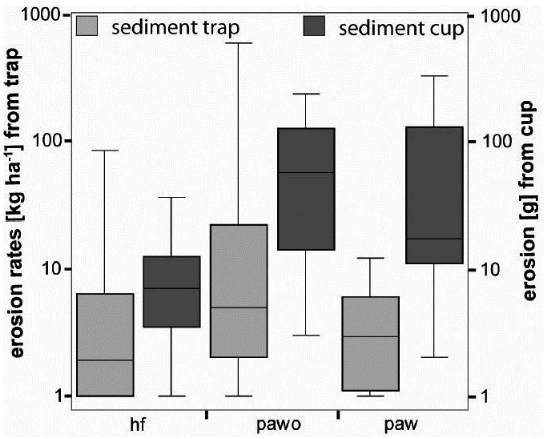
1.2 Quantification of sheet erosion rates – a challenging task in (sub-)alpine areas

1.2.1 Soil erosion estimates derived from field measurements

Soil erosion measurement with sediment traps

In the Urseren Valley, monthly sediment yields recorded with sediment traps during the growing seasons of 2007 and 2008 ranged from 0 kg ha^{-1} (two hayfield sites in September 2007) to 580 kg ha^{-1} (pasture in August 2007; Fig. 4) (Konz et al. 2012). These rates can be considered as very low especially when compared to the visible damages. This is even more surprising since the average fractional vegetation cover was reduced for all grassland types considered: with 87.2% in the hayfield sites (hf), 64.6% in the pasture sites (pawo), and 76.3% in the pastures with dwarf shrubs (paw). Soil erosion rates measured with sediment traps did not significantly differ between single grassland types within the observed period ($p > 0.05$, ANOVA).

Fig. 4 > Monthly soil erosion measurements with sediment traps and sediment cups in dependence on the land-use type



Konz et al. 2012

One possible explanation is that the measurement equipment was only installed during the growing season. Furthermore, no erosive rain events occurred during the measurement period of 2007–2008. The measured erosion rates are congruent with the ones measured during the growing season by Felix and Johannes (1995), which were 0.1 to 200 kg ha^{-1}. The authors concluded that their low erosion rates were based on low effective precipitation (rainfall that contributes to surface runoff), which was estimated at between 1 and 2%. This is also true for the investigated plots. Effective precipitation estimated from the measured surface runoff of the plots ranged from 0.6 to 2% during the growing seasons of 2007 and 2008.

In contrast to these results, Frankenberg (1995) measured a mean (for six years) erosion rate of 20 t ha^{-1} yr^{-1} (during the growing season) on grassland test plots with clear signs of degradation (reduced vegetation cover). These high rates might be due to an effective precipitation during the investigation period of up to 60%.

Soil erosion measurement with sediment cups

Since soil erosion rates during the growing seasons were low and possibly transport limited, sediment cups were installed to test whether there is small-scale soil movement within the plots that is not captured, due to the occurrence of sedimentation before the material reaches the sediment traps. Sediment cups are only suitable for a qualitative assessment of relative differences in erosion activity and cannot provide information on soil erosion rates, since the source area of the sediment is not known. The sediment cups indicated a comparable erosion activity for pastures with and without dwarf shrubs but a lower activity for hayfields. The latter might be explained by small-scale soil detachment and transport due to trampling. The high erosion activity indicated by sediment cups points to small-scale soil movement in which detached particles are not transported down-slope and are thus not captured by sediment traps. Other authors also found that generally 70 to 85% of the eroded material remains near the point of detachment (Walling 1983).

The slight relative differences in erosion rate between land-use types is in agreement with the results of several sprinkling experiments conducted in the Bavarian Alps and in South Tyrol, which showed an increase in runoff and erosion on grassland in the following order:

1. Covered vegetation area without grazing or other mechanical impact.
2. Covered vegetation area with additional impact (cultivation, grazing).
3. Areas without vegetation cover (Schauer 1981; Bunza 1989; Markart et al. 2000).

Soil erosion measurement with ^{137}Cs and $^{239–240}Pu$

Compared to the low soil-erosion estimates during the growing season, soil erosion rates based on in situ NaI measurements of ^{137}Cs were considerably higher: 6 to 37 t ha^{-1} yr^{-1}. In contrast to the sediment traps, the ^{137}Cs method integrates all erosive processes that occur since the time of the radionuclide's fallout. As such it provides a long-term annual soil erosion rate. Mean ^{137}Cs activity based on in situ measurements was 91 Bq kg^{-1} (SD ±19%) for all hayfields, 94 Bq kg^{-1} (SD ±27%) for all pastures, and 121 Bq kg^{-1} (SD ±27%) for all pastures with dwarf shrubs. The resulting erosion rates were consequently highest for the hayfields, resulting in an average erosion rate of 27 t ha^{-1} yr^{-1}, followed by the pastures (25 t ha^{-1} yr^{-1}), and considerably lower rates were found for the pastures with dwarf shrubs (8 t ha^{-1} yr^{-1}). For the two hayfield sites with high values, high snow-glide rates were also observed and were likely largely responsible (beside mouse activity) for the observed erosion rates. The lowest erosion rates were found for pasture with dwarf shrubs. The latter might be due to increased sedimentation induced by the dwarf shrubs, which act as physical barriers that reduce the transport of soil particles down-slope (as described above). Moreover, they also act as a barrier between the snow and soil interface. As long as no uprooting occurs, dwarf shrubs may reduce snow gliding or act as a protective layer between the sliding snow and the soil. In consistency with this argumentation, measured average snow-glide rates in the winter of 2009/2010 were highest for the hayfields, at 121 cm, while an intermediate value of 77 cm was observed for the pastures and the lowest value of 31 cm for pastures with dwarf shrubs. The first field measurements (winter 2012/13) of sediment concentration in the snow accumulation of these sites showed that the maximum erosion rate due to snow gliding was 22.8 t ha^{-1} yr^{-1} while the average rate was 1.4 t ha^{-1} yr^{-1}. High erosion rates due to snow movement were also observed in the Aosta Valley, Italy, where the estimated snow-related soil accumulation in the deposit area was 3.7 t ha^{-1} and 20.8 t ha^{-1}. The impact on snow gliding will be discussed in more detail in the following chapter.

In the Urseren Valley the very high erosion rates could be confirmed with the radionuclide $^{239+240}Pu$. These radionuclides have integrated net soil redistribution since the start of the atmospheric nuclear weapons tests in the 1950s–1960s. Moreover, $^{239+240}Pu$ contamination in our Alpine valleys originates mostly from nuclear bomb fallout. Thus, plutonium is more homogeneously distributed than ^{137}Cs fallout from the Chernobyl accident. The coefficient of variance (CV) for reference sites was 29% for ^{137}Cs distribution in the Urseren Valley (n = 6) and 95% in the Piora Valley (n = 7) (Alewell et al. 2014). In contrast, reference $^{239+240}Pu$ values had CVs of 10 and 18% for the reference sites at Urseren Valley (n = 6) and Piora Valley (n = 7), respectively. Soil erosion assessment using $^{239+240}Pu$ as a tracer pointed to a huge dynamic and high heterogeneity of erosive processes (sedimentation of 0.9–6.4 t ha^{-1} yr^{-1} and erosion of 2.6–21.8 t ha^{-1} yr^{-1} in the Urseren Valley and sedimentation of 0.7–77 t $ha^{-1}yr^{-1}$ and erosion of 1–5.3 t ha^{-1} yr^{-1} at Val Piora) (Alewell et al. 2014).

1.2.2 Soil erosion estimates resulting from erosion risk models

Comparison of the soil erosion models WEPP, PESERA, and USLE

Many reviews of soil erosion models are available, but only a few of them discuss soil erosion models applicable to mountain regions (Thornes 2007; Begueria et al. 2008). Soil erosion modelling in mountain regions is problematic because most models were developed for cultivated, flat, or hilly areas and are not able to describe the erosion behaviour of natural vegetation or that for steep and long slopes. But the most serious limitation is that processes related to freeze–thaw cycles, permafrost, snow melt, and snow movement are usually not considered. Thus, the validity of existing models must be critically tested in mountain areas (Van Rompaey and Govers 2002). An evaluation of different soil erosion risk models for their suitability in mountain regions will be discussed in more detail in chapter 3.3.

Water Erosion Prediction Project (WEPP) estimates for alpine grassland plots

The Water Erosion Prediction Project (WEPP) is a physically based model and was used to simulate the hydrology and sediment yield of the nine erosion plots. Generally WEPP simulated the hydrology (soil moisture, surface runoff) of the sites sufficiently, even though the snow melt was delayed by one month compared to observations. The latter caused additional production of meltwater and therefore the water availability in spring and early summer was overestimated. For the WEPP application, a distinction was made between short-term erosion prediction for a single growing season (2007) and long-term erosion prediction for the period 1986–2007, which corresponds to the time span that is captured by the ^{137}Cs method. The short-term simulated erosion rates for single growing seasons were very low at 0.3 kg ha^{-1} for the hayfields and pastures and 0.9 kg ha^{-1} for pasture with dwarf shrubs, which would be less than 0.0016 t ha^{-1} yr^{-1} (Konz et al. 2010). Even though the susceptibility of the sites to soil erosion is high, with snow melt, steep slopes, and erodible silty to loamy sand textures, WEPP simulated low to zero erosion. The simulated erosion rates were even lower than the rates measured with the sediment traps. Mean long-term annual erosion rates simulated for the period of 1986–2007 ranged from 0.04 to 1.21 t ha^{-1} yr^{-1} (Konz et al. 2010). The simulated cumulative erosion rates for growing seasons (from April to October 1986–2007) are 30 to 500 times less than the annual rates from January to December (1986–

2007). The higher cumulative annual erosion rates result from simulated overland flow on frozen ground and extreme rainfall events in the period before 2007.

Pan-European Soil Erosion Risk Assessment (PESERA)

The Pan-European Soil Erosion Risk Assessment (PESERA; Kirby et al. 2004) is a physically based model with a low input data requirement. PESERA-VBA is its 2-D application and is suitable for the plot modelling. However, the model distinguishes between only six soil texture classes and 12 land-use types, which limits the differentiation of the plots. This is especially true for our sites, which differ in soil texture, land use, and fractional vegetation cover (FVC), of which only differences in FVC could be considered. The average soil erosion rate modelled with PESERA was 0.37 t ha^{-1} yr^{-1}.

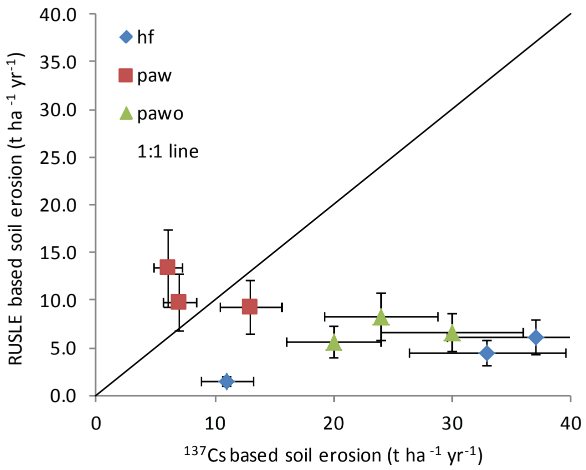
The empirical (Revised) Universal Soil Loss Equation (RUSLE) – at plot scale

The Universal Soil Loss Equation (USLE) is an index based, empirically derived model. RUSLE retains the six factors of *Agriculture Handbook no. 537* to calculate erosion from a hill slope. Technology for evaluating these factor values has been computerized and new data added (Renard et al. 1997).

The RUSLE provides long-term average soil erosion rates. The simulated values differ depending on the grassland type. Mean average erosion rates are lowest for the hayfields at 3.6 ±1.9 t ha^{-1} yr^{-1}). Estimates for pastures are higher at 10.5 ±3.7 t ha^{-1} yr^{-1}) and 8.3 ±2.0 t ha^{-1} yr^{-1}) for pastures without dwarf shrubs (Fig. 5). The higher values for pastures are caused by a lower fractional vegetation cover and thus a higher C-factor. Results of the RUSLE on the land-use type pasture with dwarf shrubs are of a similar order of magnitude as the ^{137}Cs-based erosion rates if the uncertainty of methods is considered (Fig. 5). The uncertainty of the USLE is based on the measurement error of each single parameter: slope angle, grain size, organic carbon, and fractional vegetation cover.

Fig. 5 > Erosion rates estimated with the ^{137}Cs-method (in situ Na-I detector) and calculated with the Revised Universal Soil Loss Equation (RUSLE)

Caesium-137 error bars (17%) are due to manual analysis of gamma spectra. RUSLE error bars are obtained by consideration of the uncertainty of single parameters.



The pastures and hayfields were underestimated by the RUSLE compared to ^{137}Cs-based erosion rates. The reason for the underestimation 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 Fig. 6, whereas no damages could be seen on plots with dwarf shrubs (Fig. 6).

These snow-related soil erosion processes are not considered within the RUSLE.

Fig. 6 > Eroded sites some days after snowmelt on hayfield hf1 (left-hand side) and on pasture without dwarf shrubs pawo2 (right-hand side) in the Urseren Valley

Both pictures illustrate visible erosion damages after the winter time.

Konz et al 2009

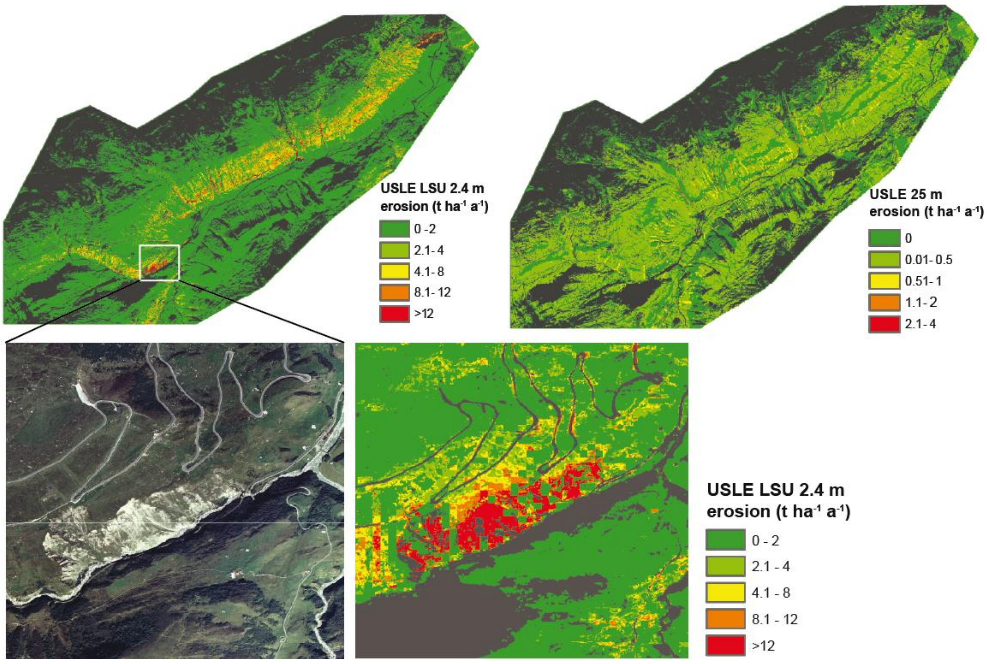
RUSLE – a catchment-scale application

The RUSLE is one of the few models that can be applied on regional and national scales due to its simplicity and rather low data demand. However, published RUSLE estimates for the alpine areas deviate considerably. For example Friedli et al. (2006) reported RUSLE-based values of 0–6 t ha^{-1} yr^{-1} while a prediction of the Joint Research Centre with RUSLE gives a range of 0–50 t ha^{-1} yr^{-1} (Bosco et al. 2008) for the Swiss Alps. The PESERA model estimates an even smaller erosion risk of <1 t ha^{-1} yr^{-1} for the Alps (Kirkby et al. 2008). These large discrepancies can mainly be attributed to different parameterization of the vegetation cover.

The extent to which soil erosion estimates may differ assuming an intact 100% grassland cover and by applying the actual mapped values of fractional vegetation cover was tested for the Urseren Valley.

Fig. 7 > Estimated soil loss (t ha^{-1} yr^{-1}) by RUSLE calculated with the low resolution and uniform C factor map (based on the Swisstopo land-cover dataset) and the C-factor based on the QuickBird-derived actual fractional vegetation cover data

Bottom: a visible comparison between the high-resolution RUSLE soil erosion map and the pan-sharpened QuickBird image. The land-cover types shadow, snow, and rock are displayed in black.



Meusburger et al. 2010b

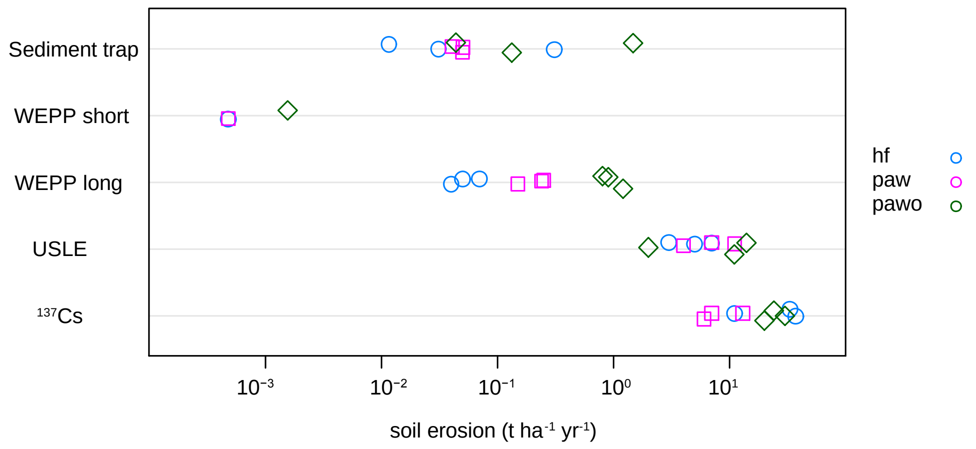
Using the uniform, intact vegetation cover, maximum soil erosion values are about 2.8 t ha^{-1} yr^{-1} and the mean modelled erosion rate for the entire catchment is 0.08 t ha^{-1} yr^{-1} (Fig. 7, right). The highest values are found along the channels in the catchment and mainly display the pattern of the water flow accumulation. The areas in between are characterized by very low erosion rates of <0.5 t ha^{-1} yr^{-1}. The erosion estimates produced by considering the actual, often reduced vegetation cover range from 0.0 to 16 t ha^{-1} yr^{-1} with a mean erosion of 1.18 t ha^{-1} yr^{-1} for the catchment (Fig. 7, left). High erosion rates are concentrated on the south-facing slopes, where grassland cover is scarce due to pasturing and shallow landslides (Meusburger and Alewell 2009). The highest values occur at spots of bare soil located along rivers, roads, and landslides (Fig. 7, lower part). The visual evaluation as well as the comparison to ^{137}Cs erosion estimates indicates that the results of the actual soil erosion risk are more plausible considering the frequently fractional vegetation cover of alpine grassland.

1.3

Comparison of soil erosion rates for different processes and scales

Because the different erosion assessment tools used in this study captured different erosional processes (Tab. 1), they delivered a wide range of erosion rates (Fig. 8). Especially the deviation between the sediment traps and the ^{137}Cs method is high.

Fig. 8 > Soil erosion rates ($t\ ha^{-1}\ yr^{-1}$) yielded by different approaches



Using the sediment traps, several processes were neglected such as snowmelt, snow-movement, and extreme rainfall events (mainly because during the measurement window 2007/2008 no extreme rainfall events occurred). The snow melting processes can further increase erosion rates, as shown by the long-term WEPP application. However, WEPP did not account for the soil detachment by trampling. And thus the erosion rates are on average comparable to the sediment traps.

The long-term estimates of RUSLE are closer to the average annual rates measured with the ^{137}Cs method. The RUSLE estimates were higher because its empirical factors are based on long-term measurements and, in this study, the R, L and S factors were adapted for mountainous areas. Use of empirically based parameters produced more plausible long-term soil erosion estimates but it does not consider winter and snowmelt processes and is thus still expected to underestimate soil erosion rates. In fact, only the ^{137}Cs approach integrates all erosive processes that occur over a long period and as such it is more likely that extreme rainfall events will also be included.

Neglecting snowmelt and the mechanical impact of snow results in a reduction of the erosion rate by 100% for our sites. Further, the neglect of trampling by WEPP or the lack of extreme events for the sediment traps causes a further significant reduction of erosion rates. Consequently, in the Urseren Valley erosion processes during the growing season seem to have a minor influence on annual soil erosion rates, in the absence of extreme events. The deviation between the ^{137}Cs and the other approaches is most likely due to winter processes, most of all the mechanical translocation by snow movement (gliding, ablation, and avalanches) (Meusburger et al. 2013a).

Tab. 1 > Overview of different processes captured by the two soil erosion models and the two measurement techniques

Method	WEPP short	WEPP long	(R)USLE	PESERA	Sediment trap	^{137}Cs, $^{239+240}Pu$
Sheet erosion growing season	√	√	√	√	√	√
Trampling, grazing	X	X	√	X	√	√
Extreme events	X	√	√	X	X	√
Snow melt	X	√	X	√	X	√
Mechanical impact of snow	X	X	X	X	X	√
Average soil loss (t ha^{-1} yr^{-1})	>0.002	1.5	10	>0.4	1.5	20, 10

Taken together, erosion values based on commonly used methods like sediment traps and soil erosion modelling with WEPP and PESERA imply low erosion risk. Also the RUSLE model shows very low values if intact uniform vegetation cover is assumed. However, if the actual vegetation cover is implemented, RUSLE is able to successfully estimate an erosion rate in a similar range, as was found by measurements made during the growing season in different areas of the Alps.

Erosion rates reported from other alpine areas

Different measurements on test plots across the Alps showed, in agreement with our sediment trap measurements, that an intact vegetation cover prevents almost all soil loss (Felix and Johannes 1995; Frankenberg et al. 1995; Langenscheidt 1995). However, on grassland plots with clear signs of degradation (reduced vegetation cover), erosion rates can be considerably higher. For example a mean erosion rate of 20 t $ha^{-1}yr^{-1}$ (during a six-year measurement period) was observed by Frankenberg et al. (1995) on plots located on flysch and molasse material in the Allgäuer Alps. In the Bavarian Alps (Kalkalpen), Felix and Johannes (1995) found erosion rates of 4.4 t $ha^{-1}yr^{-1}$ (during a two-year measurement period) on a pastured grassland test plot with a fractional vegetation cover of 66%. In another region of the Bavarian Alps, Ammer et al. (1995) measured soil erosion rates of approximately 2–9 t ha^{-1} yr^{-1} (during a five-year measurement period) after clear cutting of the small forested catchments, which geologically belong to the flysch and Kalkalpen formations. A review of erosion measurements on marls in the French Alps names erosion rates of 14 to 33 t ha^{-1} yr^{-1} (Descroix et al. 2001). And in the northern French Alps, sediment deposits of 0.6 to 1.8 cm during single events depending on vegetation type and fractional vegetation cover have been measured (Isselin-Nondedeu and Bedecarrats 2007).

In addition to potentially high erosion rates caused by disturbed vegetation cover, locally we can observe very high erosion rates due to snow movement (avalanches and snow gliding), which has not been considered in any soil erosion risk assessments so far (Jomelli and Bertran 2001; Ceaglio et al. 2012). As a consequence we believe that soil erosion risk is generally underestimated for Alpine areas. This conclusion is further underpinned if we consider:

> the high rainfall erosivity (discussed in more detail in chapter 2.2.1)
> the pronounced relief
> the fact that the predominantly silty sands are connected to high erodibility even though erodibility is lowered by high organic matter contents and surface stone cover.

2 > Causes and risk factors for alpine soil loss

2.1 Shallow landslide susceptibility factors

2.1.1 Factors affecting natural landslide susceptibility: topography, geology, hydrology, and soil

The equation for the critical altitude for slopes

Countless studies have investigated the causes of landslide susceptibility in mountain areas all over the world. Most of them identified slope and geology as the most important determinants of landslide susceptibility (Carrara et al. 1991; Rickli et al. 2001; Dai and Lee 2002; Zhou et al. 2002; Ohlmacher and Davis 2003; Santacana et al. 2003; Van Westen and Lulie Getahun 2003; Suezen and Doyuran 2004; Ayalew and Yamagishi 2005; Clerici et al. 2006; Komac 2006). This natural susceptibility can be physically explained with the equation for the critical altitude H_c for slopes (Carson 1971):

$$H_C = \frac{4c \sin\alpha \cos\phi}{\gamma \times \left(1 - \cos(\alpha - \phi)\right)} \quad \textit{(Equation 1)}$$

where
c = *cohesion,*
γ = *bulk density,*
α = *slope, and*
Φ = *friction angle.*

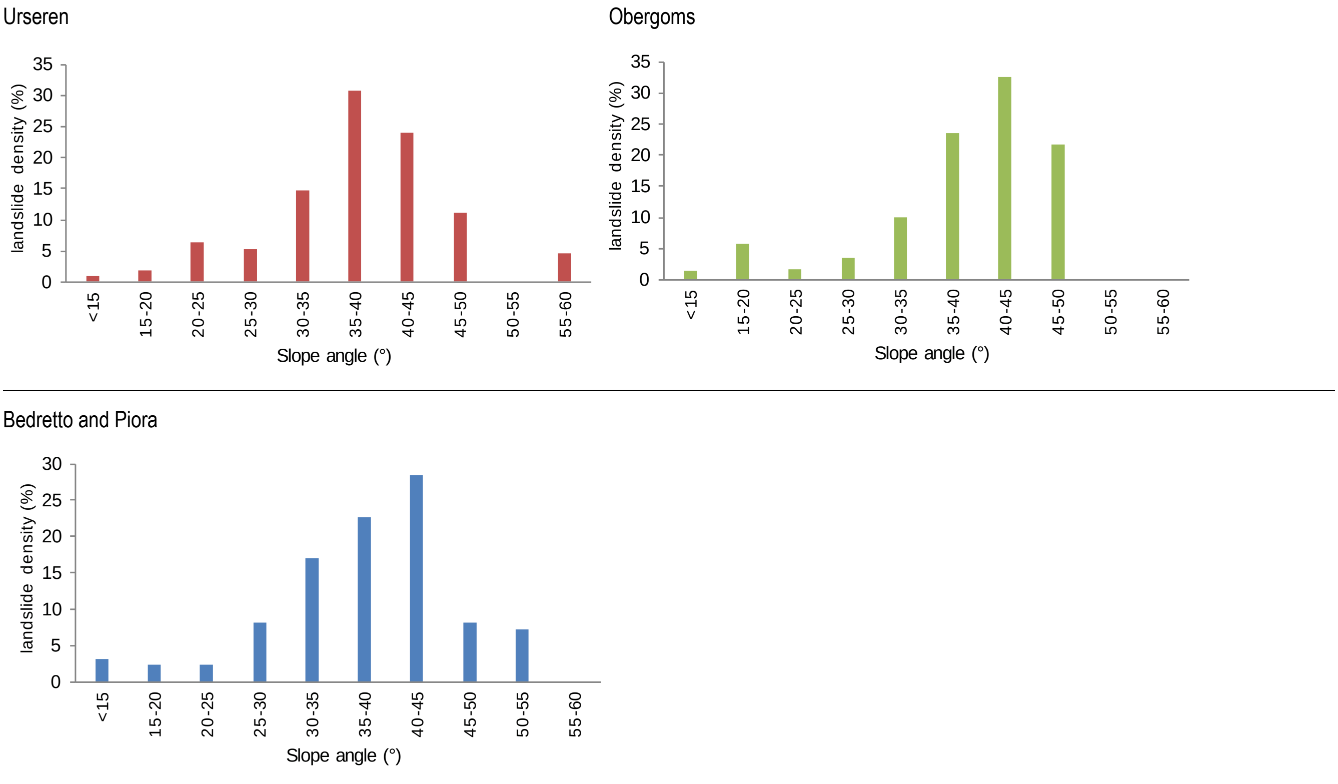
Equation 1 describes slope stability as a function of soil strength (cohesion and friction angle) and shear forces (density and slope). A slope/soil becomes instable if the actual height/thickness exceeds the critical height. The higher the slope angle and the greater the soil depth, the higher the shear force and the less stable the slope.

The soil strength parameters are influenced by parent material, which is represented by geology in many studies. Geology is often the most important factor since it not only determines material properties but also affects runoff generation and water flow in general. The balance between soil strength and shear force is a function of soil water content (Ahnert 2003), because positive pore water pressure reduces cohesion of soil particles (c), and increases soil weight and consequently the bulk density (γ).

Causal factor topography for the case study sites

For the three investigated valley areas the probability of landslides increased with slope angle and peaked at angles ranging between 35 and 45° (Fig. 9). This slope angle corresponds well with literature values for other Alpine grasslands (Dommermuth 1995; Rickli et al. 2001; Tasser et al. 2003). In general, landslide susceptibility increases with increasing slope angle and soil depth (Carson 1971). In the field, however, the landslide density decreases again for steeper slopes, most likely because soil cover becomes thinner through continuous sheet erosion. A thinner soil cover is less susceptible to landslides because lower gravitational forces occur.

Fig. 9 > Landslide density histogram depending on slope angle



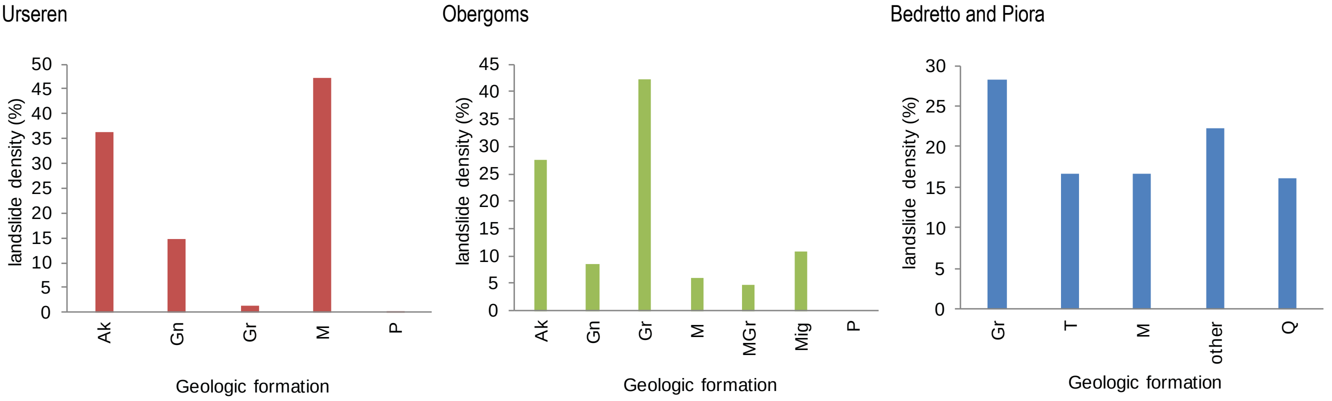
Regarding geology (Fig. 10), the three investigated valleys show different patterns. In the Urseren Valley, landslide density was highest in the Mesozoic formation and the "Altkristallin" (47.1 and 36.4%, respectively). Calcareous rock of the Mesozoic formation weathers to clay, forming soils with stagnic properties, which are particularly prone to landslides due to their layered and water-ponding character. In the Obergoms and Bedretto/Piora valleys, the Mesozoic layer is less affected by landslides (6.1 and 16.6%, respectively). In both valleys this susceptible geologic formation is largely forested. However, as soon as the protective forest cover is absent, the slopes reveal their instability (Fig. 11). Even though the Permian, like the Mesozoic, is also a sediment layer, it shows the lowest susceptibly to landslides due to its location at the flat valley floor in the Urseren and Obergoms valleys.

Causal factor geology for the case study sites

The Altkristallin, adjacent to the Tavetscher fault line, is a highly deformed and instable clay schist. This layer shows a very similar landslide density in the Urseren and Obergoms valleys. In the Obergoms Valley, the granitoids have the highest landslide densities, which is due to the location of the layer in the elevated pasture grasslands. The same is true for the granitoid layer in the Bedretto/Piora valleys.

Fig. 10 > Landslide density histogram depending on geologic formations

for the Permian (P) and Mesozoic sediments (M), granite of the Aare massif (Gr), gneiss of the Gotthard massif (Gn, "Altkristallin" (Ak), metagranitoids (MGr), migmatite (Mig), Quaternary (Q), and other formations that consist of Lucomagno, Monte Leone, Lebendun, and Maggia nappes.



Labhart 1999

Fig. 11 > Mesozoic formation without forest cover reveals instability in the Obergoms

The described interactions between different causal factors (e.g. vegetation cover and geology) need to be considered to allow the identification of the most important factors. We used multivariate analysis, especially logistic regressions, for the assessment.

Multivariate evaluation – potential causal factors involved

Several of the causal factor maps were created for each valley. Some of them are derivates of the digital elevation model (DHM25, Swisstopo), such as slope aspect (transformed using the sine and cosine representing the east versus west and the north versus south, Brenning 2009), elevation, curvature, morphologic index, and slope angle. The morphologic index is a classification of the slope and curvature map into the topographic features: peak, ridge, pass, plane, channel, or pit. Further derivates of the slope and aspect map are the flow direction and the flow length, which is the distance along a flow path. The flow accumulation is based on the number of cells flowing into each cell in the output raster. In addition, the topographic wetness index was used, which is defined as ln (α/tanβ), where α is the local upslope area draining through a certain point per unit contour length (here the flow accumulation) and β is the local slope. Further, catchment height, catchment area, catchment slope (all calculated with multiple flow direction algorithm; (Freeman 1991), and convergence indices, calculated with 10 and 50 m radii respectively to represent morphological changes on different scales, were assessed.

The VECTOR25 dataset of Swisstopo offers vector data of rivers, roads, and land-cover. It is based on a 1:25 000 map (position accuracy 3–8 m) of 1993. Continuous raster maps of river and roads were obtained with ArcGIS distance and density functions (500-m moving window). Point density and line density are used to calculated the quantity that falls within the identified neighbourhood (here 500 m) and divide that quantity by the area of the neighbourhood. ArcGIS density functions were also used to obtain a raster map of avalanche density. The avalanches that occurred since 1695 (data source: Swiss Federal Institute for Snow and Avalanche Research) were averaged for each pixel with a 500-m moving window to generate the avalanche density map. For the Bedretto/Piora valleys, a snow glide map modelled with the SSGM by Leitinger et al. (2008) was also included.

The factor map geology was created based on the definition of geologic formations by Labhart (1999) and refined by field and air photograph mapping for the Obergoms and Urseren valleys. Thus, for the lower accessible formations (Mesozoic, Permian), the mapping scale could be improved from 1:200 000 to 1:25 000. In addition, for the Urseren Valley a geomorphologic map was generated based on a Quaternary map with a scale of 1:33 000 (Fehr 1926) and the air photographs. Tectonic fault lines of the Swiss geological map (1:500 000) were transferred to a raster map by calculating the distance from a fault line. The same map was used for the Bedretto/Piora valleys, since no detailed map covering the entire region was available.

The precipitation map used is based on long-term (1961–1991) mean precipitation data (*Hydrological Atlas of Switzerland*, Swiss Federal Office for the Environment). Inverse distance weighted (IDW) interpolation was used to create a map. For the Obergoms and Bedretto/Piora valleys, the RUSLE R-factor map was used additionally as a predictor (Meusbuger et al. 2012). The RUSLE R-factor is the product of the kinetic energy of a rainfall event and its maximum 30-minute intensity (Brown and Foster 1987) and is the rainfall characteristic most strongly correlated to sediment yield.

For the Urseren and Obergoms valleys, pasturing maps were variable. The present and past land uses of the Urseren Valley were determined with pasture maps of the years 1955 and 2006 (Russi 2006) that were digitized, georeferenced, and rasterized. The land use in the Obergoms valley in 1967 could be established with the Alpkataster reports for the communities of Oberwald, Ulrichen, and Obergesteln. The actual land use was mapped by a farmer and proved by aerial photographs and field trips. The number of large stock units per pastured hectare for 100 days (called *Normalstoss*, which we translate into “grazing intensity”) was assigned to the pasture areas based on the direct payments. As a result, quantitative changes in grazing intensity per pasture could be established (termed “land-use change map” in the following) for the Obergoms Valley.

Landslide susceptibility modelling

Stepwise logistic regression (Allison 2001) was adopted to find the best-fitting model describing the relationship between the dependent variable and a dataset of independent variables. As the dependent variable, the presence or absence of landslides was chosen. Prior to multivariate analysis, collinearity diagnosis to prove the independency of the predictor variables is essential. For each valley, several cross-correlations were identified. For instance, an inter-correlation between elevation and land cover was observed, which was mostly due to the relationship whereby the higher the area, the less grass and the more rock surfaces were present. However, there was no causal relationship between landslide occurrence and elevation. Hence, the elevation was excluded from further analysis.

A stepwise forward variable selection based on the Akaike Information Criterion (AIC) was used. The AIC measures the goodness-of-fit while penalizing model complexity to obtain a model that explains the occurrence of landslides almost as well as a more complex model (Akaike 1974). Smaller models with fewer predictors keep the estimated coefficient standard errors small and prevent overfitting (Hosmer and Lemeshow 2000). Overfitting runs the risk that an algorithm or model will perform very well on the available training data to which it is fitted but poorly on new test data and will therefore produce unreliable predictions (Hosmer and Lemeshow 2000). The setup of the logistic regression models was accomplished based on the landslide inventory maps beginning in the year 2000 and independent causal factor maps of each valley. The most recent landslide inventory maps were kept for temporal validation of the model. A balanced sample of one-third of the instable and stable cells was chosen for the training of the model. The remaining two-thirds of the cells were used for the spatial validation of the model.

Significant causal factors identified by the model

Seven out of nine independent variables were included after the model generation with the stepwise forward selection method in the Urseren Valley due to a significant explanation of the variance by geology, slope, stream-density, road-density, avalanche density, distance to roads, and flow accumulation (Meusburger and Alewell 2009).

Slope and avalanche density were the most significant predictors in the Obergoms ($p < 0.0001$), followed by slope aspect and geologic formation ($p < 0.001$) and finally land-use change (Tab. 2).

Tab. 2 > Estimates for multiple logistic regression coefficients, related standard error, and significance (Sig.) for the independent parameters in the Obergoms

Predictors	Estimate	Std. error	Sig.
Intercept	1.22E+00	1.51E+00	
Avalanche density	1.57E-01	3.23E-02	***
Catchment area	–1.47E-04	1.41E-04	
Flow convergence 50 m	7.73E-05	4.86E-04	
Flow convergence 10 m	–6.01E-03	5.27E-03	
Curvature	–2.41E-01	6.31E-01	
Geologic formation	–1.32E-01	4.26E-02	**
Land-use change	–6.34E-01	3.12E-01	*
Land-use class	4.36E-02	6.52E-02	
Precipitation	–1.69E-03	8.92E-04	°
Road density	6.54E-05	8.81E-05	
Cos slope aspect	–1.21E-05	3.87E-06	**
Slope	2.68E+00	3.74E-01	***
Stream density	–6.55E-02	5.36E-02	

Significance codes: *** = 0.001; ** = 0.01; * = 0.05; ° = 0.1; 1

In the Bedretto/Piora valleys, several predictors have a significant influence on the landslide distribution. Most significant are slope aspect, the R-factor, and slope. Of minor significance are the predictors avalanche density and geologic formation and the hydrologic parameters flow length and stream density (Tab. 3). Against expectation, the landslide susceptibility decreases with increasing R-factor. The R-factor decreases with elevation because the proportion of precipitation falling as snow increases. Since in the Bedretto/Piora valleys the landslides predominantly occur on the elevated sites, a negative relationship between R-factor and landslide susceptibility is the result.

Landslide susceptibility is dependent on different predictors in each valley. In the Urseren Valley the results seem to confirm the widely recognized view that geology greatly influences the occurrence of landslides as it determines the strength and permeability of rock as well as the resultant cohesion of the soil layer (Carrara et al. 1991; Guzzetti et al. 1999; Dai and Lee 2002). Slope was confirmed as a very important predictor in all three valley areas, as was avalanche density. The latter is interesting since it indicates that landslides in alpine areas are triggered not only by rainfall but also by snow movement. Another indication of the importance of snow movement in landslide triggering is the significance of slope aspect in the Obergoms and especially the Bedretto/Piora valleys. However, the snow-glide map produced with the SSGM by G. Leitinger which was available for the Bedretto/Piora valleys did not significantly increase the predictive ability of the model.

Tab. 3 > Estimates for multiple logistic regression coefficients, related standard error, and significance (Sig.) for the independent parameters in the Bedretto/Piora valleys

Predictors	Estimate	Std. error	Sig.
Intercept	5.54E-01	1.70E+00	
Avalanche density	5.70E-02	1.74E-02	**
Catchment area	–2.73E-07	5.19E-07	
Catchment height	2.93E-04	9.20E-04	
Flow convergence 50 m	–6.82E-03	4.73E-03	
Flow convergence 10 m	4.63E-03	3.83E-03	
Flow length	–1.05E-01	3.32E-02	**
Geologic formation	1.37E-01	4.25E-02	**
Precipitation	1.61E-03	1.04E-03	
R-factor	–1.89E-03	3.91E-04	***
Road density	–4.20E+00	1.58E+01	
Snow-glide distance	1.30E-04	7.18E-05	°
Slope	1.50E+00	4.24E-01	***
Stream density	–3.86E+01	1.91E+01	*
Cos aspect slope	–6.74E-01	1.14E-01	***

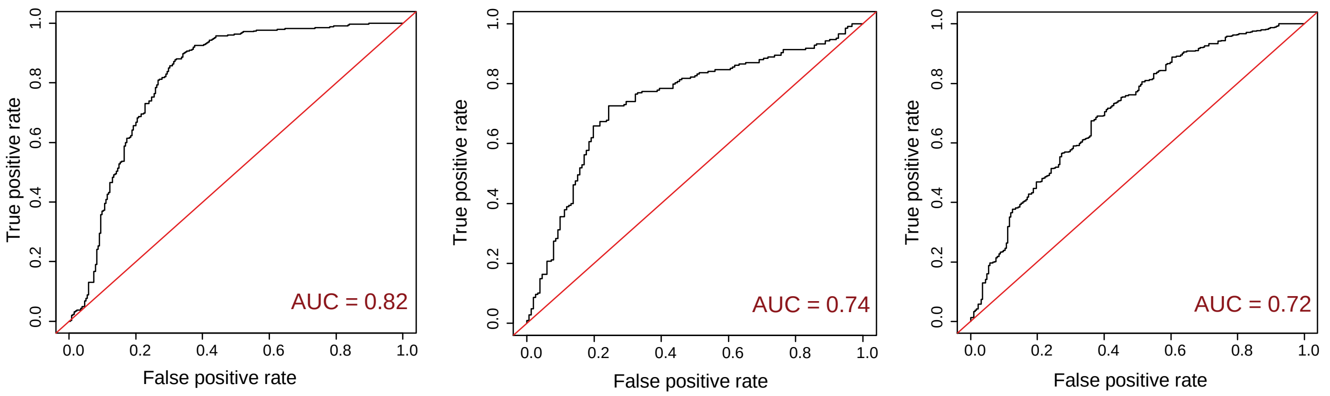
Significance codes: *** = 0.001; ** = 0.01; * = 0.05; ° = 0.1; 1

Performance of the landslide susceptibility models

A common performance measure is the success rate (Chung and Fabbri 2003, 2008). With the significant set of predictors (presented above), success rates (for the training dataset) of 81.4, 72.7, and 67.0% were achieved for the Urseren, Obergoms and Bedretto/Piora valleys, respectively. For the validation dataset the performance of the model was still good for the Urseren Valley at 74.5% and moderate for the Obergoms Valley at 61.7%, while for the Bedretto/Piora valleys the discrimination was not acceptable (55.6%).

The quality of the calibrated models was further evaluated with the Area Under the ROC Curves (AUC). ROC curves plot "sensitivity" versus 1-"specificity", where sensitivity is the proportion of grid cells containing known landslides that are correctly classified as susceptible, and specificity is the proportion of grid cells outside a mapped landslide that are correctly classified as stable (Swets 1988; Lasko et al. 2005). The higher the curve above the diagonal line (i.e. AUC = 0.50), the better the model. The AUC values give congruent results with the best landslide susceptibility mapping (highest AUC) being achieved for the Urseren Valley and the lowest for the Bedretto/Piora valleys (Fig. 12).

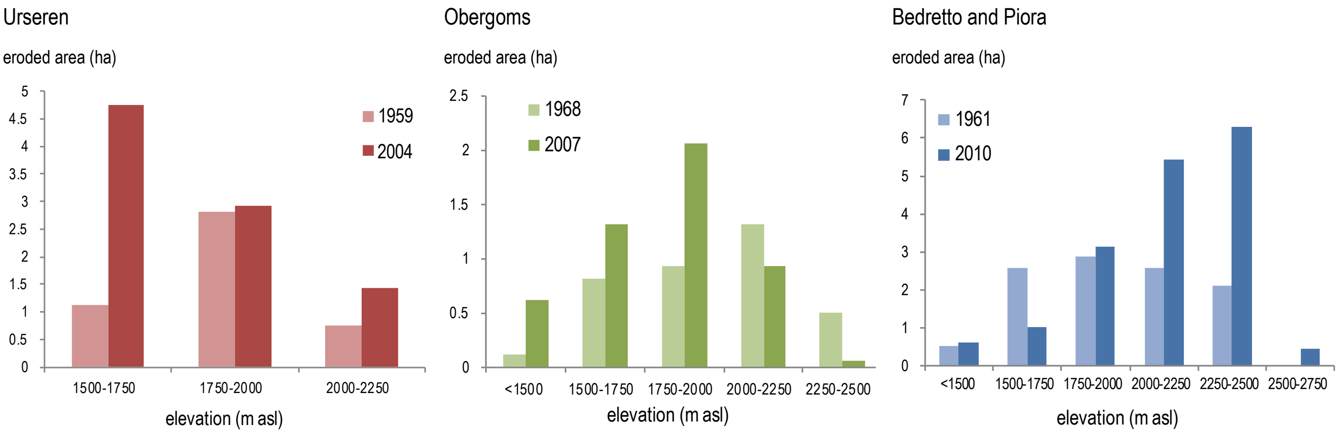
Fig. 12 > ROC curves with AUC values for the training dataset of the Urseren, Obergoms, and Bedretto/Piora valleys (from left to right)



2.1.2 Factors related to trends in landslide susceptibility: climate and land use

For all three investigated valley areas, a distinct increase in landslide occurrence over time was observed (Fig. 3). However, the spatial pattern of the increase is very different. In the Urseren Valley the increase occurred mainly on the lower south-facing pastured slopes close to the valley floor. In the Obergoms Valley the region of the alp pastures was most affected by the increase, since the lower geologically susceptible slopes are mostly forested. The affected areas are partly subject to slight intensification and partly subject to abandonment since the Alpkataster report in 1967. Both of these land-use developments are expected to increase landslide susceptibility. The increasing effect of intensification could be shown for the Urseren and Obergoms valleys. However, an increase in the shallow landslide occurrence could not be observed for abandoned sites (Fig. 13). Published data point to an increased impact of snow movement on the vegetation cover of abandoned pastures. In the Bedretto/Piora valleys the increase occurred at altitudes above the managed grassland. Thus, it is likely that climate factors such as changes in snow cover and maybe also melting of permafrost are predominant driving factors.

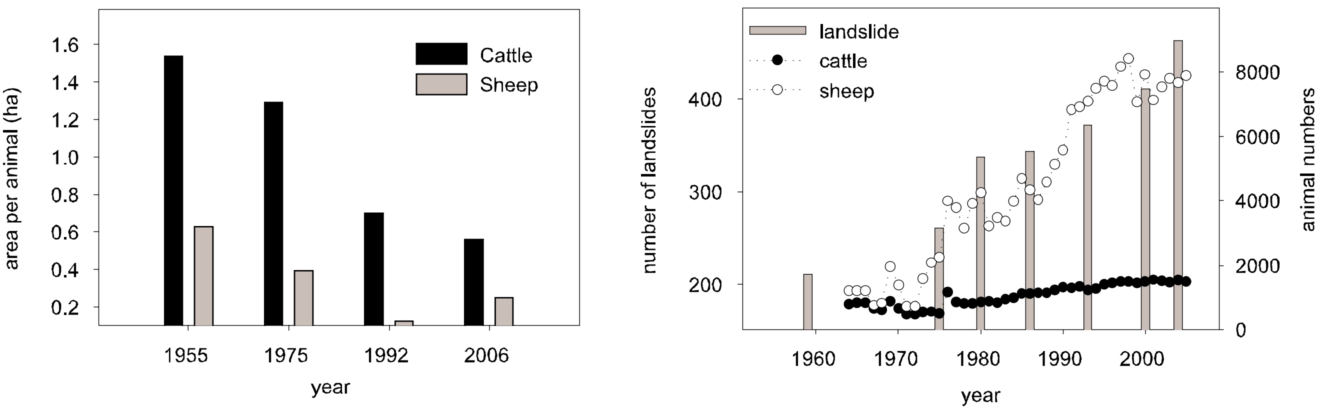
Fig. 13 > Shift in landslide susceptibility zones observed during the last five decades for the three sites: Urseren, Obergoms and Bedretto / Piora valleys



Effects of land-use intensification – example of Urseren Valley

In the Urseren Valley, most severe changes in land use took place in the beginning of the 1970s as local agriculture became mechanized and traditional farming practices were abandoned. Moreover, land use was mainly intensified in the valley during the last decades, which is shown by the decreasing pasture area per animal (Fig. 14). After 1955, the pasture space per cow decreased steadily, mainly due to an increase in cow numbers from 785 to 1482 and a reduction of cow pasture area. Pasture area per sheep also decreased until 1992 due to an increase in the number of sheep from 1193 in 1955 to 7875 in 2006. The sheep pastures were enlarged in 2006. Goats are of minor importance in the valley and their number decreased from approximately 550 in 1976 to 280 in 2006.

Fig. 14 > Land-use intensification for sheep and cattle (left) and a comparison between the increased landslide and stocking numbers (right)



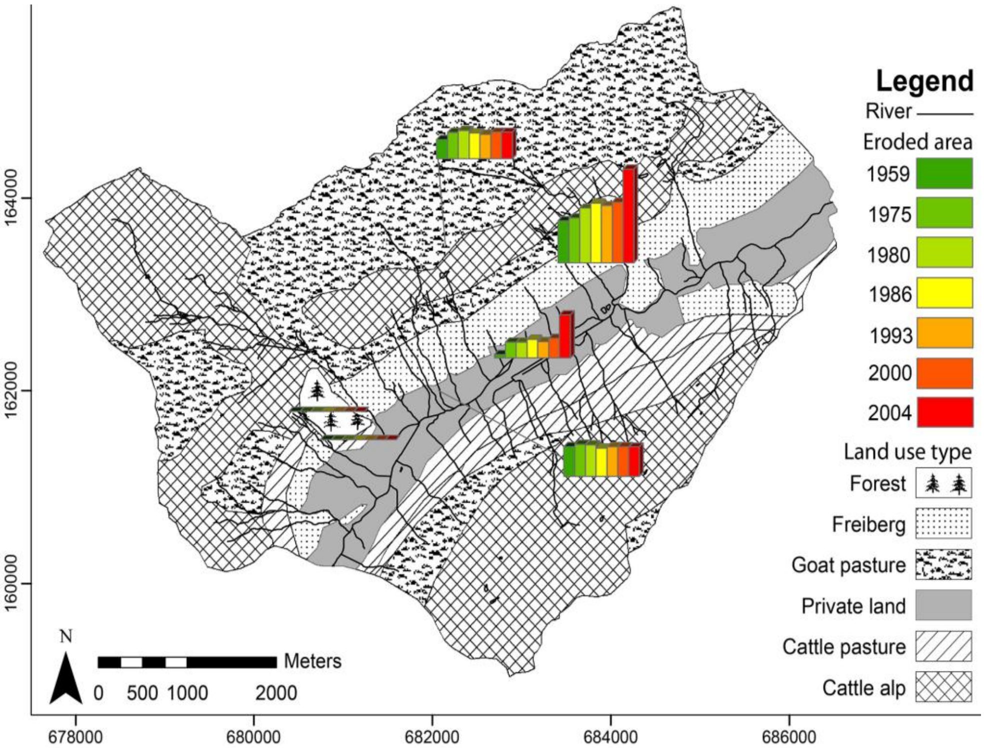
Meusburger and Alewell 2008

The spatial pattern of the increase in landslide occurrence is clearly related to land use. The display of landslide areas for single years for each traditional land-use type of 1955 (Fig. 15) shows that the remote and extensively used pastures were already slightly affected by landslides in 1959, but no increase in landslides could be observed over time. Today these areas are almost exclusively used as sheep pastures.

The intensified areas closer to the valley floor (the former *Freiberg* and private land) have clearly destabilized. *Freiberg* areas are pastures which are used in spring because of the vicinity to the farms and the more advanced vegetation state at this altitudinal level. The appointed date on which the cattle are brought to the higher pastures is 14 June. For the rest of the summer the *Freiberg* is kept as a reservoir in case of early onset of winter and is left to regenerate during the main growing season. The *Freiberg* zone was already the most affected zone in 1959 due to its vulnerable geology, whereas the private land at that time was almost undisturbed.

To summarize, we can see a clear impact of the changed management practice in the Urseren Valley: The accessible, more intensively used areas destabilized, whereas areas of extensification to the point of abandonment did not destabilize further.

Fig. 15 > Progression of eroded area by landslides on different traditional land-use types of 1955

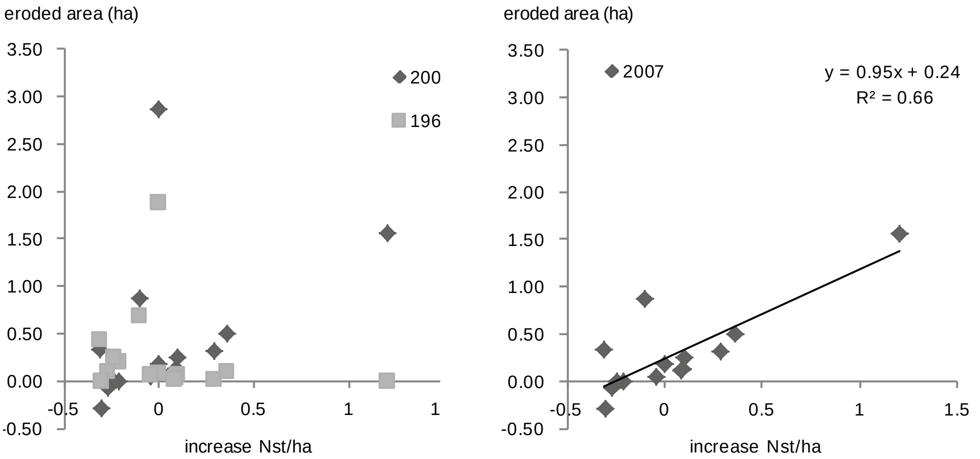


Meusburger and Alewell 2008

Effects of land use intensification and extensification – example of the Obergoms Valley

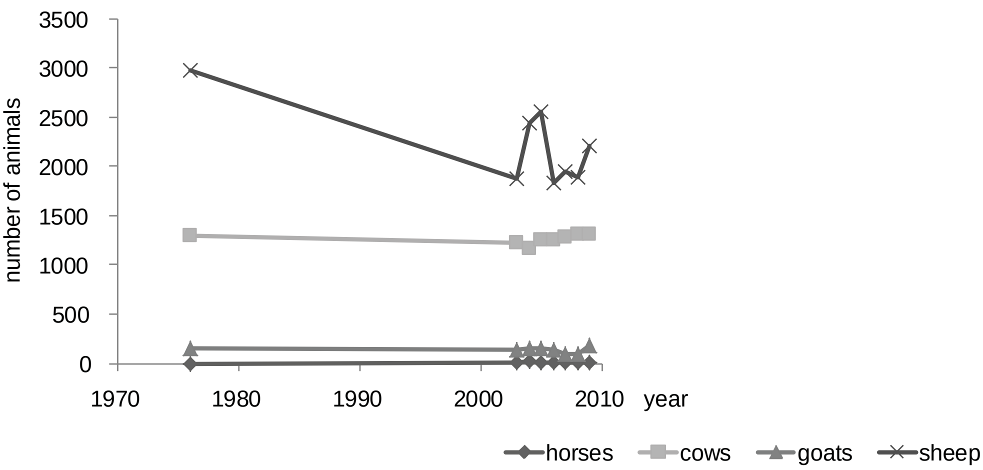
Increases in landslide incidence are observed for both intensified areas and areas that have never been used (see rectangle in Fig. 16, left) in the Obergoms Valley. One of the most interesting aspects of the data is the increase in destabilized areas that have never been managed. These areas were already heavily affected by landslides in 1968, because they are usually steeper than the managed areas. Thus, they have a higher natural susceptibility to landslides that was further enhanced during the last four decades, which might indicate that changes in climatic conditions triggered the higher landslide susceptibility. However, the climatic conditions did not affect all areas equally. Without considering the areas that have never been managed, a correlation between change in grazing intensity and area eroded by landslides is evident (Fig. 16, right). With increasing grazing intensity (expressed as change in normal stocking unit per hectare), the eroded area increases. However, the correlation is driven by one high value; for the negative values, indicating extensification, the corresponding landslide susceptibility is highly variable. The relationship between grazing intensity and landslide susceptibility is most likely non-linear, with heavy overgrazing triggering strong responses in landsliding. The strong effect that land use may have on landslide susceptibility is for instance visible for the pasture that was most strongly intensified. Before the intensification in 1968 the area was not affected by landslides at all; in 2007 it was the area most affected by landslide erosion. On the extensively managed grassland, both increases and decreases in landslide susceptibility could be observed (Fig. 16).

Fig. 16 > Change in pasture intensity versus eroded area for the years 1968 and 2007 (left) and only for the year 2007 (right), excluding the areas that have never been managed



In the Bedretto Valley, land use extensified slightly due to a decline in the number of pastured sheep (Fig. 17). In the Piora Valley where only cattle are pastured the land use did not change. In both valleys the number of pastured cattle remained constant since the mid of the 1970s and the number of sheep decreased from approximately 3000 to 2000, with large variability in the last decade (Fig. 17). Goat pasturing has always been of minor importance. In total the number of large stock units declined from 2363 in the year 1963 to 1843 in the year 2008. However, the change in pastured normal livestock units (from 1474 to 1363) is marginal. The discrepancy is explained by more pasturing days per animal and growing season nowadays. Whereas in the Obergoms Valley the pasturing area declined, the managed area in the Bedretto/Piora valleys remained constant and is still based on the alp separation contract of the year 1227. For these almost constantly slightly extensified land-use areas no significant change in landslide susceptibility was evident since the time of the first aerial photos in 1959 and 1961.

Fig. 17 > Development of number of different pasture animals in the Bedretto/Piora valleys



Land-cover: abandonment and shrub encroachment

Parallel to the abandonment of grassland management, alteration of land-cover occurs, which manifests in the encroachment of shrubs. The invasion of shrubs is observed for the whole sub-alpine belt of the Swiss Alps. An increase of almost 11.8% was assessed for *Alnus viridis* between 1983/85 and 1993/95 (Brassel and Brändli 1999).

In the Urseren Valley the database allowed the mapping of shrub encroachment since 1959. Land cover is neither an independent parameter (it is usually co-linear to other parameters) nor a significant one for landslide susceptibility modelling. Nonetheless, to evaluate the effect of shrub encroachment on the landslide trend, landslide densities are presented for sites stocked pre-1959 and for those which have been invaded since 1959 (Tab. 4).

Tab. 4 > Landslide densities for different land-cover types and areas that have been invaded by shrubs since 1959 (new shrub)

Land cover	Landslide density
Forest	0.0
Debris	7.5
Shrub after 1959	2.9
Shrub before 1959	33.2
Grassland/dwarf shrub	56.4

The shrub cover showed considerable change over time. The area covered by shrubs increased since 1959 by 30.0%. The landslide density is 33.2% for the community of old shrubs but only 2.9% for the new shrubs. The older shrub cover mainly occurs in places typical for *Alnus viridis*, that is, on sites with wet conditions, like tributary channels on the north-facing slope. These wet and steep areas close to channels are especially susceptible to landslides. The new shrub cover, however, occurred in untypical places for *Alnus viridis* (Wiedmer and Senn-Irlet 2006) on the plains between the channels and on the south-facing slope. The invasion is most likely caused by inhomogeneous grazing. These recently invaded areas show low susceptibility to landslides. The cause of the low landslide susceptibility of the new shrub category can probably be explained by the two other functions of vegetation. *Alnus viridis* is often used in bioengineering to mechanically stabilize slopes with its roots (Graf et al. 2003). In addition, the high evapotranspiration rates of *Alnus* species (Herbst et al. 1999) and the affinity to wet conditions are effective in regulating the soil water budget (Wiedmer and Senn-Irlet 2006). However, we cannot directly deduce a stabilizing effect of *Alnus viridis* from our data. The highest landslide densities could be observed for the vegetation types of dwarf shrub and grassland (56.4%). Debris, with a landslide density of 7.5%, is relatively less susceptible to landslide activity, as the inhomogeneous mixture of materials and grain sizes stabilizes the soils (Ahnert 2003). The small forested area (0.7% of the catchment) is not affected by landslides.

Extensive investigations have shown that vegetation cover, especially of the woody shrub type (such as *Alnus viridis*, *Sorbus aucuparia*, *Acer pseudoplatanus*, *Pinaceae*, and *Fagaceae*) helps to improve the stability of the slopes (Schauer 1975; Gray and Leiser 1982; Greenway 1987; Rickli et al. 2001; Zhou et al. 2002; Graf et al. 2003).

Generally, vegetation has three effects on slope stability: (i) mechanical anchoring of the soil, (ii) regulation of the soil water budget (iii), and mechanical resistance to snow or snow gliding.

Snow gliding can cause fissures in the soil cover that are potential tear-off lines for landslides (Schauer 1975; Newesely et al. 2000). In the investigated valleys, *Alnus viridis* constitutes the dominant species within the shrub category. *Alnus viridis* was less affected by landslides than the grassland/dwarf-shrub category, which consists mainly of pastures with high proportions of *Rhododendron ferrugineum*, which were found to stabilize the snow cover (Newesely et al. 2000).

Snow gliding for different vegetation cover types

Measurement of snow glide rates for the different vegetation types in the winter of 2009/2010 could confirm lower snow glide rates for the *Alnus viridis* sites compared to grassland sites (Tab. 5).

Tab. 5 > Parameters related to snow-glide distance (sgd) for the investigated sites in the Urseren Valley, Switzerland

"N" indicates the sites on the north-facing slope.

site	vegetation	slope (°)	initial force Fr (g m s^{-2})	static friction coefficient µs (-)	measured sgd (cm)
h1	hayfield	39	569	0.37	189
h2	hayfield	38	510	0.33	50
h3	hayfield	35	392	0.24	126
pw1	pasture with dwarf-shrubs	38	1 030	0.66	34
pw2	pasture with dwarf-shrubs	35	1 118	0.69	28
p	pasture	38	579	0.37	89
p	pasture	35	1 109	0.68	64
h1N	hayfield	28	343	0.20	30
h2N	hayfield	30	608	0.35	8
pN	pasture	18	628	0.33	17
A1N	*Alnus viridis*	25	1 050	0.58	2
A2N	*Alnus viridis*	30	451	0.26	28
A1	*Alnus viridis*	22	1 550	0.84	14
A2	*Alnus viridis*	31	1 197	0.70	60

The link between snow-glide rate and vegetation cover is given by the static friction coefficient. The static friction coefficient can be derived by:

$$\mu_S = \frac{F_r}{F_n} \qquad \textit{(Equation 2)}$$

where
F_n = *the normal force, which can be calculated with*

$$F_n = m \times g \times \cos\alpha \qquad \textit{(Equation 3)}$$

where
g = *the gravitational constant (9.81 ms^{-1}),*
α = *the slope angle (°), and*
m = *the weight of the snow-glide shoe (in our study 202 g).*

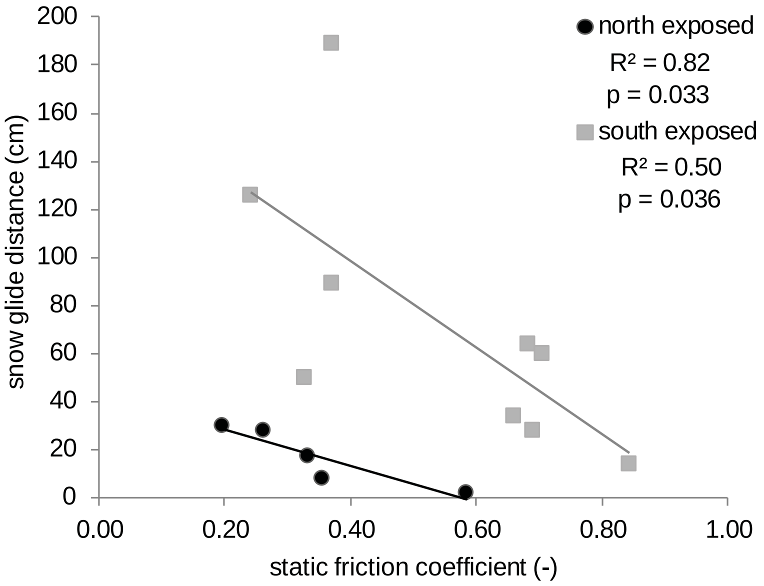
The initial force (Fr) is needed to get the glide shoe moving on the vegetation surface.

The lowest surface roughness was observed for the hayfields, followed by sites covered with *Alnus viridis* on the north-facing slope (Tab. 4). For the pastures with dwarf shrubs, the two mean monitored values differed (μ_s = 0.4 and 0.7) but were similar to those of pastures without dwarf shrubs (μ_s = 0.6 to 0.7). Slightly higher values were observed for the dense undergrowth of *Alnus viridis* sites on the south-facing slope (μ_s = 0. 7–0.8).

The measured snow-glide distances of the different sites varied from 1 to 189 cm (Tab. 4). The main proportion of this variability can be explained by the slope exposition and the surface roughness (Fig. 18). With increasing surface roughness (expressed as the static friction coefficient; μ_s) the snow glide distance declines. This decrease is more pronounced for the south-facing slope ($y = -1547.2\mu_s + 172.93$; $R^2 = 0.50$; $p = 0.036$). For the north-facing slope the snow-glide distances are lower. Approximately 80% of the observed variability on the north-facing slope can be explained by the surface roughness ($y = -622.17\mu_s + 43.09$; $R^2 = 0.82$; $p = 0.033$). For the south-facing slope, the snow glide distances are higher and only 50% of the variability can be explained by the difference in surface roughness. The identification of slope exposition and surface roughness as the main causal factors for snow gliding correspond to the findings of other studies (In der Gand and Zupancic 1966; Newesely et al. 2000; Hoeller et al. 2009).

Our measured snow-glide distances are comparable to those recorded by other researchers. For example, during a seven-year period in the Austrian Alps, Höller et al. (2009) monitored a snow-glide distance of 10 cm within the forest, 170 cm in cleared forest sites, and up to 320 cm in open fields. Margreth (2007) found total glide distances of 19 to 102 cm for an 11-year observation period in the Swiss east Alps (southeast-facing slope at 1540 m a.s.l).

Fig. 18 > Snow-glide distance against the static friction coefficient for the south- (squares) and north- (dots) facing slope sites



Land-use management affects snow movement

The data show a clear link between vegetation cover and snow gliding. This is of main interest in the context of abandonment. The effect of abandonment and the subsequent succession states on slope stability have been discussed in detail (Schauer 1975; Karl 1977; Bunza 1989; Newesely et al. 2000; Tasser et al. 2003). The fact that a reduction of the agricultural use enhances landslide risk is ascribed to increased snow gliding on these areas (Schauer 1975). Other factors are the higher vulnerability of the succession states and less maintenance of the sites by farmers (Tasser et al. 2003).

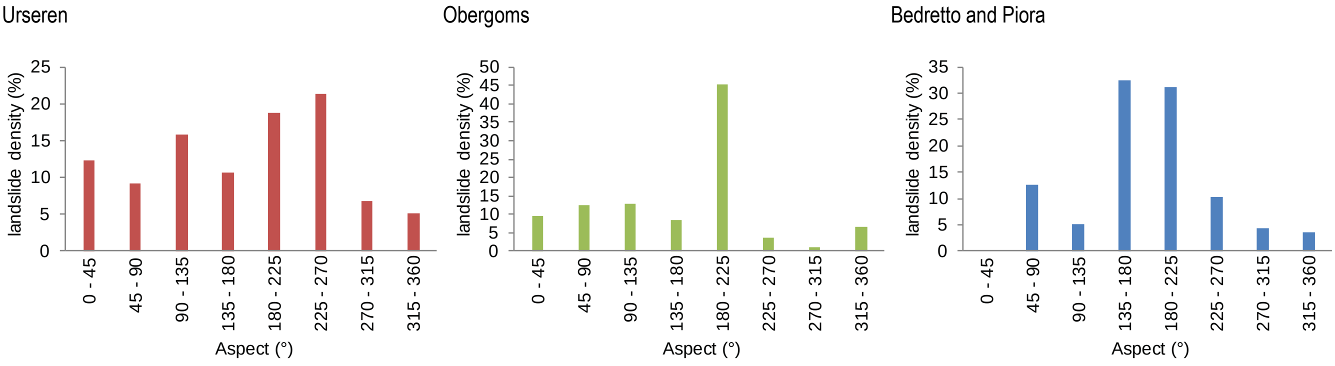
For the Urseren Valley, an increase in snow gliding on the abandoned sites was not observed. One possible explanation is that generally in the Alps, increases in landslides on abandoned land mainly occur in areas with slopes above 58° inclination (Karl et al. 1985; Mössmer 1985; Bätzing 1996). In contrast the abandoned sites are located on a relatively flat shoulder in the Urseren Valley. For the Obergoms Valley, we could observe an increase in shallow landslides on the unmanaged land, since these are also typical sites susceptible to snow gliding. In the Bedretto/Piora valleys too, enhanced landslide susceptibility cannot be attributed to abandonment. Abandonment leads to invasion of low dwarf shrubs such as *Calluna vulgaris* or *Arctostaphylos uva-ursi* (Newesely et al. 2000). These species are of great importance because they are a solid stage in the succession after abandonment of alpine ecosystems. These soft dwarf shrubs, which can be easily pressed down by the snow cover, facilitate snow gliding (Newesely et al. 2000) and may enhance landslide susceptibility. Thus, an increase in snow gliding due to changed climatic factors is likely and will be discussed below.

Effects of snow cover on landslides susceptibility

Snow processes are related to slope aspect. The latter determines the temperature and duration of the snow cover and the snow melt and movement as well as soil humidity in relation to evaporation. Thus, this parameter is highly relevant for both snow movements and the hydrologic conditions in the soil (Moser 1980; Tasser et al. 2005).

Fig. 19 shows the slope aspects where landslides predominate in the three valley sites. For the Urseren Valley no distinct pattern is distinguishable, expect for a lower susceptibility of the northwest-facing forested slopes. Landslides are concentrated on the south- to southwest-facing slopes in the Obergoms Valley and on southeast- to southwest-facing slopes in the Bedretto/Piora valleys. The same was also observed for nine Alpine sheep pastures located across the Swiss Alps (Bauer 2013). Similar distributions were found by Wiegand et al. (2010) for the Tyroler Alps, by Blechschmidt (1990) for the Karwendel, and by Laatsch and Baum (1976) for the Bavarian Alps. The observed clustering of landslides on east and southeast aspects is interpreted as being related to snow accumulation in the lee of dominant winds from the west and north. Furthermore, landslide triggering by snow movement, which is due to the radiation angle and sunshine hours in these aspects, is likely (Zweckl and Spandau 1987; Leitinger et al. 2008).

Fig. 19 > Landslide density histogram depending on slope aspect



Snow, either due to its movement on the soil surface or through meltwater production, is a triggering factor for shallow landslides. The question is now whether a change in snow cover characteristics due to climate change is responsible for the increase in landslide occurrence in elevated areas such as in the Bedretto/Piora valleys. If temperature trends are taken as a proxy for changes in snow cover characteristics, this question can be answered affirmatively. The weather stations of all three sites show increasing temperature trends with changes of 1 to 2° since the 1980s and 1950s, respectively (Fig. 20). These values are even higher than the data analyses of twelve stations in Switzerland for the period 1864–2000, for which Begert et al. (2005) found temperature rises of 1.1 and 0.6 °C per century on the northern and southern sides of the Alps, respectively. Increasing air temperature and especially the increasing number of days with air temperatures above zero influence the occurrence of snowfall and time of snowmelt and are expected to result in a rising snow line (Birsan et al. 2005).

Climate change: temperature increase

In this respect, two effects of temperature need to be considered. At lower altitudes, the decrease in snow depths and duration of snow cover (<1000–1300 m a.s.l. in January–February; (Laternser and Schneebeli 2003; Beniston 2006) might increase soil erosion from water but might decrease damages due to snow movement. At high elevations (>2000 m a.s.l.; Beniston 2006), snow depth is predicted to increase, which might result in stronger effects of snow movement.

Fig. 20 > Mean annual temperature for the weather stations in Andermatt (Urseren), Ulrichen (Obergoms), and Piotta (Ticino)

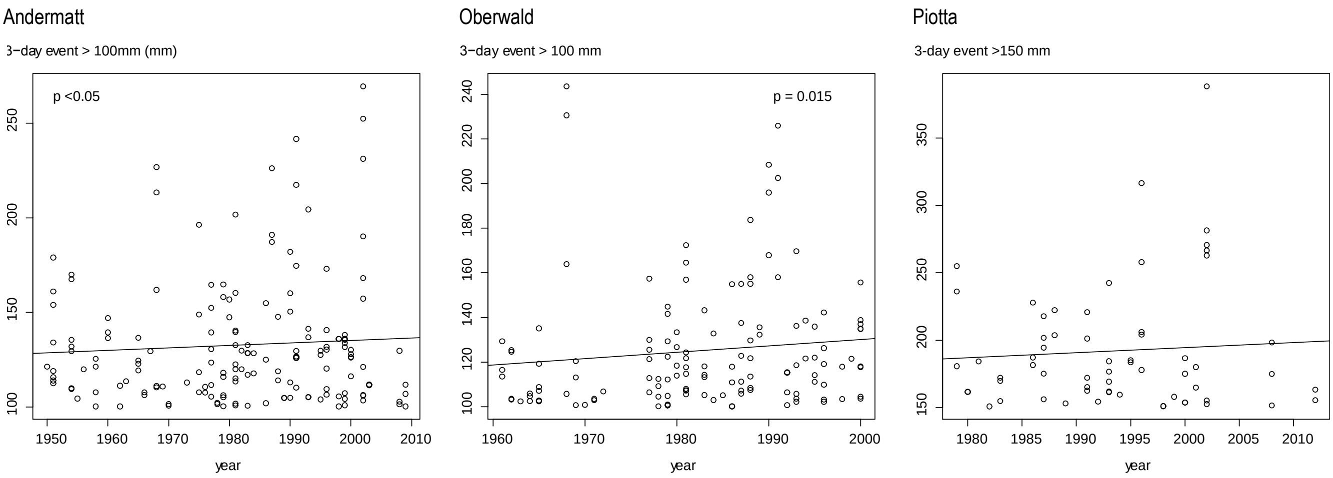
The solid lines show the linear regressions, with significant increasing trends.

Generally, landslide triggering due to precipitation is regulated by threshold functions (Zhou et al. 2002; Guzzetti et al. 2006). Farmers reported that it is mainly intense prolonged rainfall of 2–3 days that triggers landslides in the Urseren Valley (Meusburger and Alewell 2008). Three-day events exceeding a rainfall amount of 150 mm were used as a proxy for this type of event. Except for the Bedretto/Piora valleys, where only a short rainfall time series was available, we found significant increasing trends for these potentially landslide-triggering rainfall events (Fig. 21). A more detailed discussion on trends of torrential rainfall events will follow below.

Climate change: increase in intense rainfall events

Fig. 21 > Three-day rainfall events exceeding 100 m for the weather stations in Andermatt (Urseren), Oberwald (Obergoms), and Piotta (Ticino)

Solid lines show the linear regressions. Significant increasing trends are observed for the Andermatt and Oberwald stations.



2.2 Sheet erosion risk factors

2.2.1 Climatic factors: precipitation and snow

Rainfall erosivity mapping and its seasonal variability

Rainfall is one of the main drivers of sheet erosion and its relation to sediment yield is given by the rainfall erosivity, which quantifies the kinetic energy of raindrop impact and rate of associated surface runoff. Because field measurements of the kinetic energy of rainfall are scarce in both space and time, empirical relationships between conventional rainfall characteristics and soil detachment have been established, of which the most prominent and widely-used for temperate zones is the RUSLE R-factor. The R-factor is defined as the sum of all erosive events during a one-year period (Wischmeier and Smith 1978; Brown and Foster 1987; Renard et al. 1997).

Only a few studies exist that have determined the R-factor directly from high temporal resolution rainfall data in mountain areas of Europe (Rogler and Schwertmann 1981; Strauss and Blum 1994; Loureiro and Coutinho 2001; Mikos et al. 2006; Angulo-Martinez et al. 2009). For Switzerland the rainfall erosivity map is so far based on a combined approximation equation proposed by Friedli et al. (2006), where (i) the long-term R-factor is approximated by average annual precipitation (mm) after Rogler and Schwertmann (1981) and (ii) the proportion of snowfall is approximated by elevation (m a.s.l.) after Schüpp (1975). Since approximation equations have only limited transferability we aimed to evaluate the temporal as well as the spatial distribution of rainfall erosivity and to produce a map of average annual rainfall erosivity for Switzerland using the original equation by Wischmeier and Smith (1978). The only adaptation of the code is to implement temperature data in order to account for snowfall in elevated alpine areas of Switzerland (Meusburger et al. 2012).

Rainfall erosivity: spatial variation

In Switzerland the mean R-factor of all investigated stations is 1330 MJ mm $ha^{-1}\,h^{-1}\,yr^{-1}$, with a maximum of 5611 MJ mm $ha^{-1}\,h^{-1}\,yr^{-1}$ at the Locarno Monti station in Ticino (south side of the Alps) and a minimum of 124 MJ mm $ha^{-1}\,h^{-1}\,yr^{-1}$ at the Corvatsch station in Grisons (eastern Central Alps).

The R-factor was regressed with the following equation to obtain a map for Switzerland:

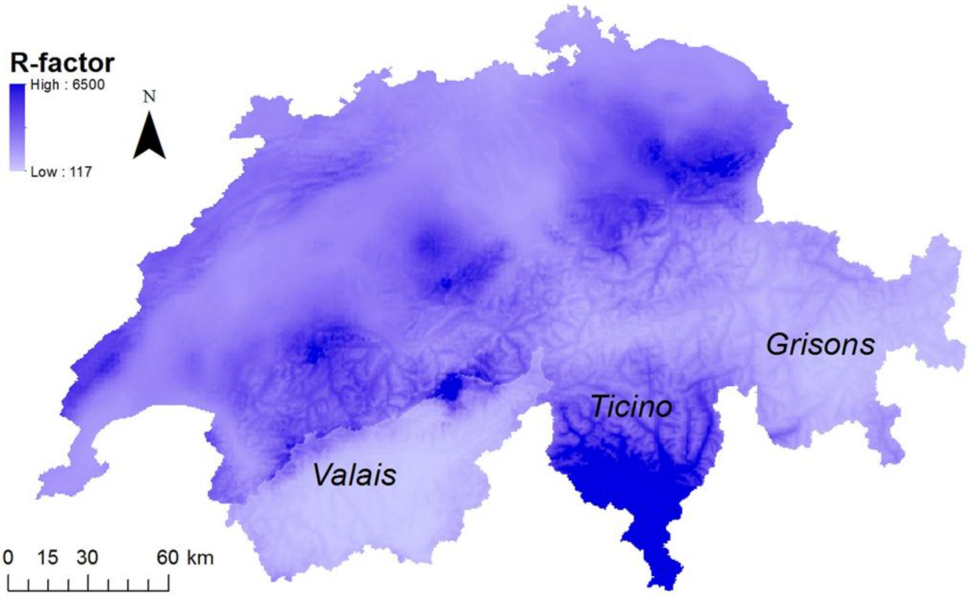
$$\log R = 0.549nP - 0.358ndem - 0.586west + 6.996 \quad \text{(Equation 4)}$$

where
nP = *the normalized average annual precipitation,*
ndem = *the normalized elevation, and*
west = *the biogeographic unit indicator map of the western Central Alps.*

The multiple regression model based on these three predictors could explain 79.5% of the observed spatial variability of the interpolation dataset and also yielded a fair prediction ($R^2 = 0.68$) for an independent validation dataset.

The descriptive statistic of the obtained Swiss R-factor map shows similar characteristics, with a mean R-factor of 1217 MJ mm ha^{-1} h^{-1} yr^{-1} that ranges from 117 to 6500 MJ mm ha^{-1} h^{-1} yr^{-1}. In general, Ticino is the region with the highest rainfall erosivity, followed by elevated stations north of the Alps (e.g. Säntis, Adelboden; Fig. 22). Medium-level erosivities are observed in the northwestern part of Switzerland. The regions with the lowest rainfall erosivities are Valais and Grisons (Fig. 22).

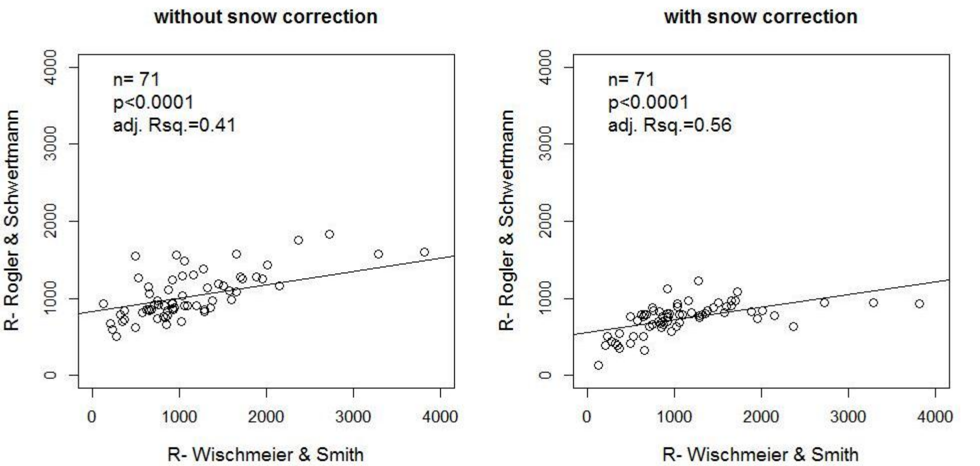
Fig. 22 > (R)USLE R-factor map (MJ mm ha^{-1} h^{-1} yr^{-1}) of Switzerland resulting from regression interpolation of 71 stations



The average R-factor values found for Switzerland are similar to the ones published by Mikos et al. (2006) for Slovenia, which range from 1580 to 2700 MJ mm ha^{-1} h^{-1} yr^{-1} for one station in different years. However, the Ticino values are more than twice as high as the maximum value observed for the Slovenian station, which can be explained by higher annual precipitation means (>1700 mm compared to 1370 mm in Slovenia) and by the strong influence of orographic rainfall in Ticino. The low erosivity values in the western and eastern parts of Switzerland are mainly due to a very low annual precipitation in combination with a high proportion of snowfall (identified by the temperature-controlled snowfall threshold). The observed R-factor pattern corresponds well to the distribution of thunderstorms in central Europe (van Delden 2001). The frequency of thunderstorms is reported to be high in the Jura Mountains, the Swiss plateau, and the Po valley at the foot of the Alps. The latter is due to the source of warm moist air of the Mediterranean sea in combination with orographic uplift, especially in the late summer season (van Delden 2001). A relative minimum of thunderstorm frequency is reported for inner Alpine valleys (van Delden 2001).

Comparison of R-factor based on 10-minute and annual average precipitation data

Fig. 23 > Comparison of R-factor calculated from 10-minute data versus R-factor based on average annual rainfall using the approximation equation of Rogler and Schwertmann (1981)

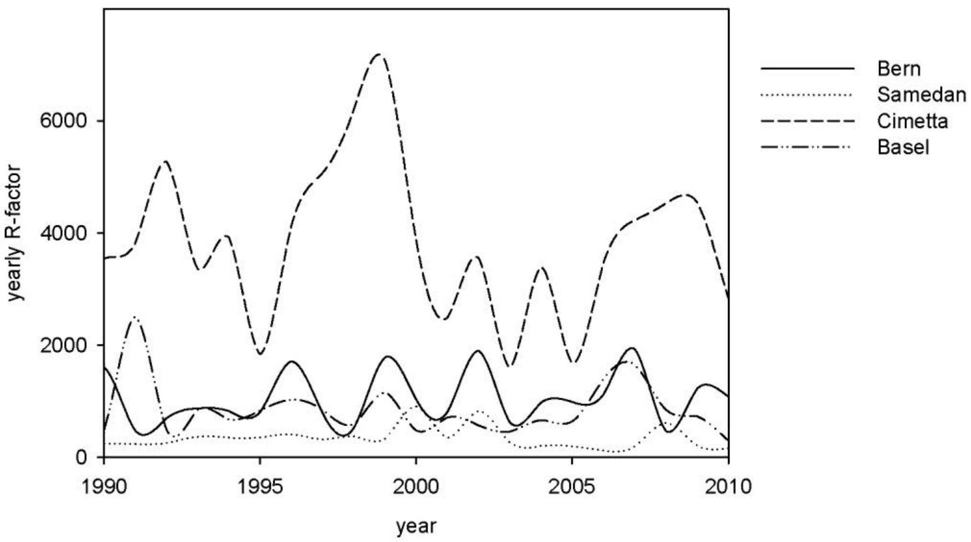


The former map after the approximation equation of Rogler and Schwertmann (1981), which was adapted to Switzerland by Friedli et al. (2006) and Prasuhn et al. (2007), yields moderate agreement (R^2 adj. = 0.56; Fig. 23, right). The original Rogler and Schwertmann equation, which does not account for snowfall, shows even lower agreement (R^2 adj. = 0.41; Fig. 23 left). Even though the spatial pattern corresponds well, the rainfall erosivity is generally underestimated: the average R-factor for all Swiss stations is 775 MJ mm ha^{-1} h^{-1} yr^{-1}, which is 42% lower than the R-factor determined using the original equation with the high temporal resolution data (10 minutes) compared to annual average precipitation data. The highest underestimation occurs for stations with high rainfall erosivity, particularly in the Ticino (e.g. Locarno/Monti, Magadino/ Cadenazzo, Stabio, etc.). A slight overestimation is observed for stations in the west and Central Alps (e.g. Visp, Grimsel Hospitz, Altdorf, and Sion). The equation of Rogler and Schwertmann (1981) was developed in the Bavarian Alps (Germany) and seems to have limited transferability to the Ticino and Valais regions due to the strong influence of orographic effects.

Rainfall erosivity: temporal variation

The resulting annual R-factor for the period 1989 to 2010 showed distinct interannual variability (Fig. 24) with coefficients of variation of, for example, 45% for Bern and 36% for Cimetta. Especially for the stations with lower R-factors, the variability is large, for example 59% for Samedan and 55% for Basel. This high variability in combination with a short time series is most likely the reason why a seasonal Mann-Kendall trend test of single stations (with data available for 22 years) could not identify significant trends in rainfall erosivity.

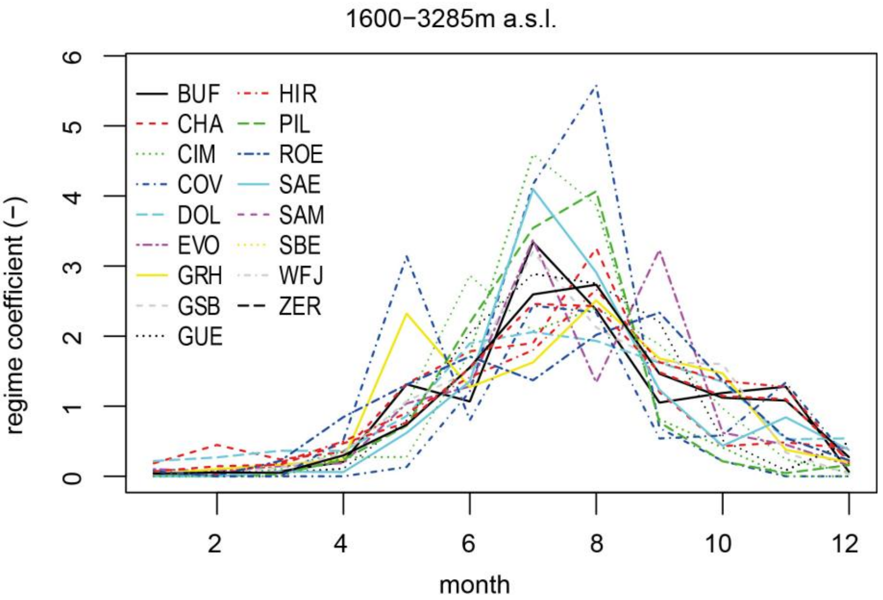
Fig. 24 > Annual variability of R-factor for stations in Ticino (Cimetta) and Grisons (Samedan) and two stations on the north side of the Alps (Bern, Basel)



The intra-annual variability of rainfall erosivity, presented as rainfall erosivity regimes, was grouped into altitudinal classes (Fig. 25). The highest values of rainfall erosivity occur in the summer months (May to September) and the lowest in the winter months (December to March). For elevated stations a more pronounced peak of rainfall erosivity is observed in summer months (three to five times higher than the average yearly rainfall erosivity), due to the long snowfall season. For most stations, rainfall erosivity peaks in July or (particularly in the Jura and the northern Alps) in August. The erosivity regimes clearly highlight the importance of monthly rainfall erosivity maps for soil erosion risk assessment (Renard et al. 1997).

Rainfall is a well-known agent of soil erosion; however, the erosive forces of snow movements are hardly considered in soil erosion risk assessments. Particularly wet avalanches can yield enormous erosive forces that are responsible for major soil loss (Gardner 1983; Ackroyd 1987; Bell et al. 1990; Jomelli and Bertran 2001; Fuchs and Keiler 2008; Freppaz et al. 2010; Ceaglio et al. 2012). Another important process of snow movement affecting the soil surface is snow gliding (In der Gand and Zupancic 1966). Snow gliding is the slow (millimetres to centimetres per day) downhill motion of a snowpack over the ground surface caused by the stress of its own weight (Parker 2002). Only a few studies investigated the effect of snow gliding on soil erosion (Newesely et al. 2000; Leitinger et al. 2008). A major difficulty for such a type of investigation is to obtain soil erosion rates caused by snow processes because avalanches and snow gliding irreversibly damage the experimental design (Konz et al. 2012).

Fig. 25 > Rainfall erosivity regimes for stations above 1600 m a.s.l.



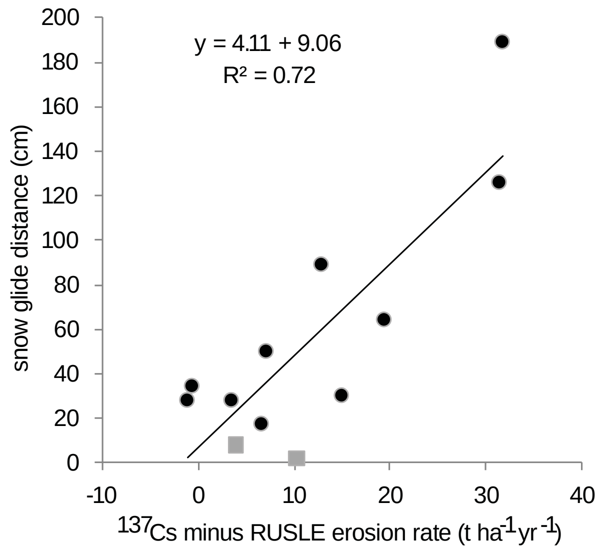
Soil redistribution related to snow movement

Besides direct quantification of sediment transport in the field by FRN-based methods (e.g. Benmansour et al. 2013; Meusburger et al. 2013b), soil erosion models (Merritt et al. 2003) can be used for soil erosion quantification. However, as mentioned above, these different methods capture different processes. The latter can also be a benefit since the comparison of the methods may make it possible to draw conclusions on the relative importance of the processes involved (see chapter 2.3).

For (sub-)alpine areas the different soil erosion processes captured by RUSLE and the ^{137}Cs method thus result in different erosion rates (Konz et al. 2009; Juretzko 2010; Alewell et al. 2014). This difference might be due to several reasons such as the error of both approaches or the unsuitability of the RUSLE model for this specific environment or it might also simply reflect the different rates of erosion processes that are captured by the two methods.

The hypothesis is that the difference between the water soil erosion rate (determined with RUSLE) and the total net erosion (determined with the ^{137}Cs method) is related to the winter soil erosion rate, which is mainly caused by snow movement. Consequently we would assume a correlation between snow movement and the difference between the two methods. To prove this hypothesis the snow movement was measured with 60 snow-glide shoes in the winter of 2009/2010 in the Urseren Valley.

Fig. 26 > Correlation of the cumulative snow-glide distances (cm) measured for the winter of 2009/2010 versus the difference between the ^{137}Cs and RUSLE soil erosion rates (t ha^{-1} yr^{-1}) for the grassland sites (dots, n = 10) and *Alnus viridis* sites (squares, n = 2)



The resulting correlation (excluding the *Alnus viridis* sites) between winter erosion and snow-glide rates was good: 73% of the variability of the winter soil erosion rate was explained by the measured snow-glide distance, with a significance of $p < 0.005$ (Fig. 26). With increasing snow-glide distance, the winter soil erosion rate also increased. The *Alnus viridis* sites, however, showed a high difference between RUSLE- and ^{137}Cs-based rates but a low snow-glide distance. A possible explanation is that ^{137}Cs was intercepted by leaf and litter material of *Alnus viridis* differently. Thus, a reference site with *Alnus viridis* stocking is necessary, but would be difficult to find in our site because flat areas are used as managed grasslands.

The observation of increasing soil erosion with increasing snow glide rates is congruent with the findings of Leitinger et al. (2008), who observed that the severity of erosion attributed to snow gliding (e.g. torn out trees, extensive areas of bare soil due to snow abrasion, landslides in topsoil) was high in areas with high modelled snow-glide distance and vice versa.

The results imply that (i) the observed discrepancies between the RUSLE- and ^{137}Cs-based erosion rates are indeed related to snow gliding and (ii) snow gliding is an important causal factor for soil redistribution in the investigated sites and probably also for other mountain sites with comparable topographic and climatic conditions.

This preliminary data indicate that low surface roughness is correlated to high snow-glide distances (Fig. 18) and these are again positively correlated to high observed differences between RUSLE- and ^{137}Cs-based erosion rates that we interpret as the winter soil erosion rate. However, the presented relation might be highly variable, depending on various factors such as soil temperature (whether the soil is frozen or not) before the first snowfall, the occurrence of a water film that allows the transition of dry to wet gliding (Haefeli 1948), the occurrence of avalanches, and the weather conditions of a specific winter.

High erosion rates due to snow movement could also be confirmed in a study site located in the Aosta Valley, Italy. There the snow-related soil accumulation estimated from the deposit area of 3.7 and 20.8 t ha^{-1} exceeded the water erosion values reported in the literature. At this avalanche-release site, these values were comparable to the yearly total erosion rates assessed with the ^{137}Cs method (13.4 and 8.8 t ha^{-1} yr^{-1}).

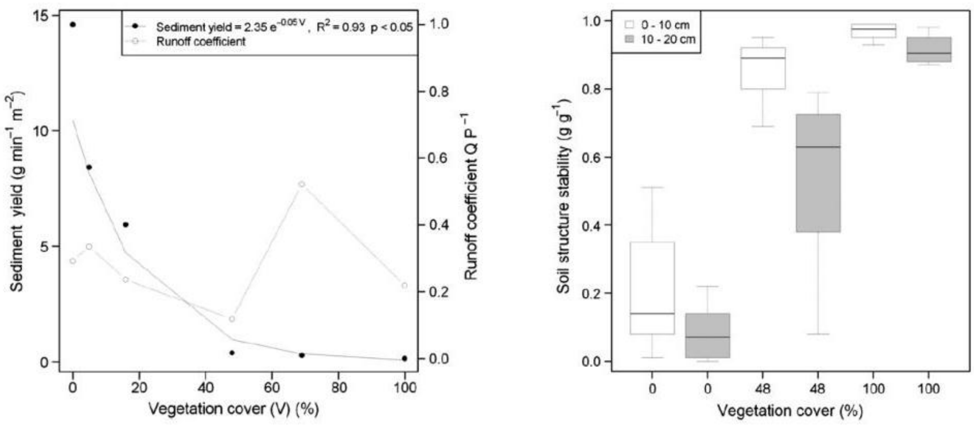
In general, the comparison of the two approaches is linked to high methodological uncertainties, mainly due to the different spatial and temporal scales considered. Still, the similarity of the erosion rates in both catchments and the snow-glide measurements indicate that soil redistribution is to a large extent driven by snow movement. As such, soil erosion processes due to snow movements should be considered in the assessment of soil vulnerability in mountain areas, as they significantly determine the pattern of soil redistribution.

2.2.2 Sheet erosion risk factors related to vegetation cover and land-use management

Importance of vegetation cover for soil erosion magnitude

Fig. 27 > Runoff coefficient (Q/P) and sediment yield per runoff in relation to the vegetation cover during a precipitation event of 2 h in the Urseren Valley (left)

Aggregate stability of soil samples from the Urseren Valley with different vegetation cover (right).



Wildhaber et al. 2012

Rainfall simulation on steep slopes in the Urseren Valley did not reveal a clear relationship between runoff and vegetation cover (Fig. 27) (Wildhaber et al. 2012). Other studies by Durán et al. (2006), Francis and Thornes (1990), Casermeiro et al. (2004), and Zhou and Shangguan (2008) found a clear relationship, while those of Martin et al. (2010) and Seeger (2007) did not. Martin et al. (2010) and Seeger (2007) explained their findings with the low vegetation height on their sites, which resulted in a low interception, and the influence of other regulating parameters like crusts, roughness, and vegetation type.

Nonetheless, sediment yield and vegetation showed a clear exponential correlation (R^2 = 0.93, p <0.05) in irrigation experiments on the slopes of the Urseren Valley ranging from sediment yields of 14.6 g l^{-1} and a total of 698.6 g m^{-2} at 0% vegetation to 0.13 g l^{-1} and a total of 2.63 g m^2 at 100% vegetation (Wildhaber et al. 2012). Several studies found a relationship between vegetation cover and sediment yield (Francis and Thornes 1990; Casermeiro et al. 2004). Differences may however occur due to different site conditions, species compositions, and biodiversity effects (Seeger 2007; Merz et al. 2009a; Martin et al. 2010).

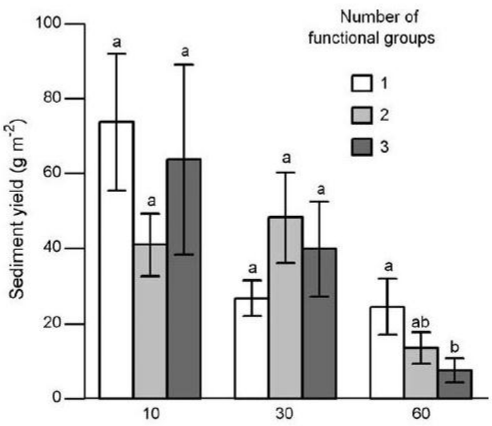
Most interestingly, soil structure stability also significantly improved with increasing vegetation coverage (one-way ANOVA, $p < 0.05$) for 0–10 cm as well as 10–20 cm soil depth (Wildhaber et al. 2012). Plants may affect aggregate and soil structure by (i) soil enmeshment and reinforcement by roots, (ii) exudation of soil-binding compounds, (iii) effects on the water balance through the soil-plant-atmosphere continuum, (iv) stimulation of microorganisms, and (v) surface protection (Tisdall and Oades 1982; Miller and Jastrow 1990; Pohl et al. 2009).

Importance of plant biodiversity for soil erosion magnitude

In another rainfall simulation study conducted in the Urseren Valley (Merz et al. 2009a), plots with differences in plant composition (herb versus grass dominance) and land-use intensity but not in plant cover (>90%) or soil conditions were investigated. Vegetation was clipped prior to the second rain-simulation to simulate intense grazing. The resulting runoff coefficient varied between 0.1 and 25%, and sediment loss was marginal, with values between 0 and 0.053 g m^{-2}, not only because of the generally high vegetation cover but also because the rainfall intensity was lower and the irrigation time shorter for this case study (Merz et al. 2009a). The data confirm the high infiltration rates of these soils and that an almost full vegetation cover prevents soil erosion even for extreme rain events. However, reduction of biomass increases soil erosion, which was shown by the clipping treatment. For this high percentage of vegetation cover, no plant-compositional effect on surface runoff or sediment yield was determined.

Fig. 28 > Sediment yield in relation to functional group diversity within different degrees of vegetation cover (10%: N = 36; 30%: N = 35; 60%: N = 35)

Means ±1 standard error. Bars which share a common letter do not differ significantly within the same class of vegetation cover (10, 30, and 60%) according to Fisher's Least Significant Difference (P <0.05).



Martin et al. 2010

For lower percentages of vegetation cover, however, plant biodiversity affected sediment yield more clearly (Martin et al. 2010). Even though the increase in functional group diversity did not decrease (as often hypothesized) sediment yield for all vegetation cover classes, the visual inspection of the data (Fig. 27) and marginally significant diversity effects at the level of single vegetation cover classes indicated a biodiversity effect. An increase in functional group diversity indeed significantly ($P < 0.05$) decreased the sediment yield at 60% vegetation cover, but did not do so at 10 or 30% vegetation cover (Martin et al. 2010). This result is also true for other diversity measures (e.g. Shannon index, species richness, species evenness). In this study of Martin et al. (2010) the functional diversity had no significant influence on surface runoff.

Effect of pasturing on soil erosion magnitude

Another important factor affecting soil erosion magnitude in grasslands is pasturing. The effect of pasturing depends on several parameters like the type of animal, weight of animal, stocking density, and pasture management (Zischg et al. 2011). Maybe the most important effect is that pasturing may decrease vegetation cover by trampling or by uprooting plants during grazing (Blankenhorn 1999; Schauer 2000), thereby reducing the soil cohesion (Blankenhorn 1999; Tasser et al. 2003). A second important process is soil compaction. Animal claws may exert a pressure of 200–500 kPa on the soil (Robertson and Scott 2009). Soil compaction can again reduce plant growth and enhance surface runoff and therefore most likely soil erosion (Bunza 1989; Sutter and Keller 2009). However, extensive farming and sustainable land-use practices increase the plant diversity (Tasser and Tappeiner 2002), which might in turn have positive effects (Martin et al. 2010).

Soil erosion rates for different land-use types

Still it is expected that negative effects of pasturing on sheet erosion predominate. For the investigated sites, soil erosion due to trampling was visible in the field (visible as clumps of soil in the sediment trap) and grain size analysis of source soils and eroded material was used to confirm the effect of trampling. The grain size distribution of eroded material in pastures is quasi-identical to that of the source soil. In contrast, eroded material from hayfields or from pastures with dwarf shrubs is finer (a high percentage of smaller grain-size classes) compared to the source soil, indicating the selective erosion process usually associated with water erosion (Konz et al. 2012).

Taken together, these studies provided evidence that a closed vegetation cover is key to reducing sheet erosion. Furthermore, the data confirm the results of catchment-scale investigations with remote sensing and modelling in the Urseren Valley which found that land-use management and percentage vegetation cover are the most important confounding factors in the assessment of soil erosion (Meusburger et al. 2010a, b). At the level of vegetation cover, however, functional diversity influenced sheet erosion. Thus, besides the establishment of stable grasslands, a potential option for soil erosion conversion measures in alpine ecosystems could be to aim at developing a more diverse vegetation cover (Martin et al. 2010).

2.3 Consequences for soil erosion assessment in the Alps

High erosion rates in the Alps may be triggered by erosive rainfalls in the summer season in areas where the vegetation cover is scarce. Thus, these parameters need to be considered together in order to identify when and where high erosion risk needs to be expected. The heterogeneity and magnitude of soil erosion rates are considerable, ranging from marginal rates for intact grasslands to >20 t ha^{-1} yr^{-1} for steep slopes affected by trampling and/or snow and freezing processes (Konz et al. 2009).

Regionalization of vegetation cover is usually done by assigning uniform values from literature or field-measured data to a classified land-cover map (Wischmeier and Smith 1978; Morgan 1995; Folly et al. 1996). The latter method results in constant values for alpine grasslands, and accounts for neither spatial nor temporal variation in vegetation cover. The use of static datasets is acceptable for parameters like soil erodibility and topography that change only slowly over time. However, vegetation cover and rainfall erosivity are characterized by a strong temporal and often seasonal variation that requires consideration. The spatial mapping of vegetation cover has improved soil erosion estimates in numerous studies (De Jong 1994; Tweddales et al. 2000; De Asis and Omasa 2007; De Jong and Jetten 2007; Meusburger et al. 2010a; Meusburger et al. 2010b). The gradual degradation of vegetation and uncovering of soil by erosion results in detectable spectral changes. When these changes are known, optical satellite data allow spatial and temporal assessment of erosion status (Shrimali et al. 2001; Liu et al. 2004).

Since rainfall erosivity is not distributed uniformly throughout the year (Mikos et al. 2006; Verstraeten et al. 2006; Meusburger et al. 2012), maps of rainfall erosivity need to be generated at least on a monthly basis. Interpolation of rainfall erosivity is challenging because of the high temporal and spatial variability; however it was successfully achieved for average long-term rainfall erosivity in Switzerland (Meusburger et al. 2012). When the resulting monthly maps are combined with maps showing the actual vegetation cover, a very good indication of risk areas is possible.

Moreover, locally very high soil removal is triggered by the movement of the snow layer in early spring. The transport capacity of the snow is very limited but may be significantly increased by subsequent snowmelt. Consequently in spring on southeast- to southwest-exposed steep grassland slopes, high erosion rates are likely if the weather conditions are favourable for snow movement and -melt.

The assessment of soil erosion related to snowmelt and snow movement is more difficult. The soil loss due to avalanches and snow gliding is locally very high. However, to date it is not possible to exactly predict when and where an avalanche will occur, which type of avalanche it will be, and how much erosive energy will be released during the event. For snow gliding, the empirical snow-glide model developed by G. Leitinger (2008) can be used to assess relative differences, but the associated kinetic energy also remains unknown. Glide distance measured with snow-glide shoes is only a proxy. These inabilities to quantify the forces that act between the snow and soil surface impede the implementation of this process in soil erosion models. New tools are needed to measure and model these snow processes related to soil erosion.

3 > Methods to assess processes of soil erosion and causal factors in alpine areas

Since most of the tools for soil erosion assessment were developed for lowland agricultural lands, method development and the identification of suitable methods for soil erosion assessment in alpine areas were a major task. The suitability of different methods from measurement to modelling and tracing isotopes was tested for the alpine environment.

3.1 Field measurements with classical methods and isotopes as indicators for soil erosion

3.1.1 Determining spring and summer sediment yield with sediment traps, sediment cups, and rainfall simulation

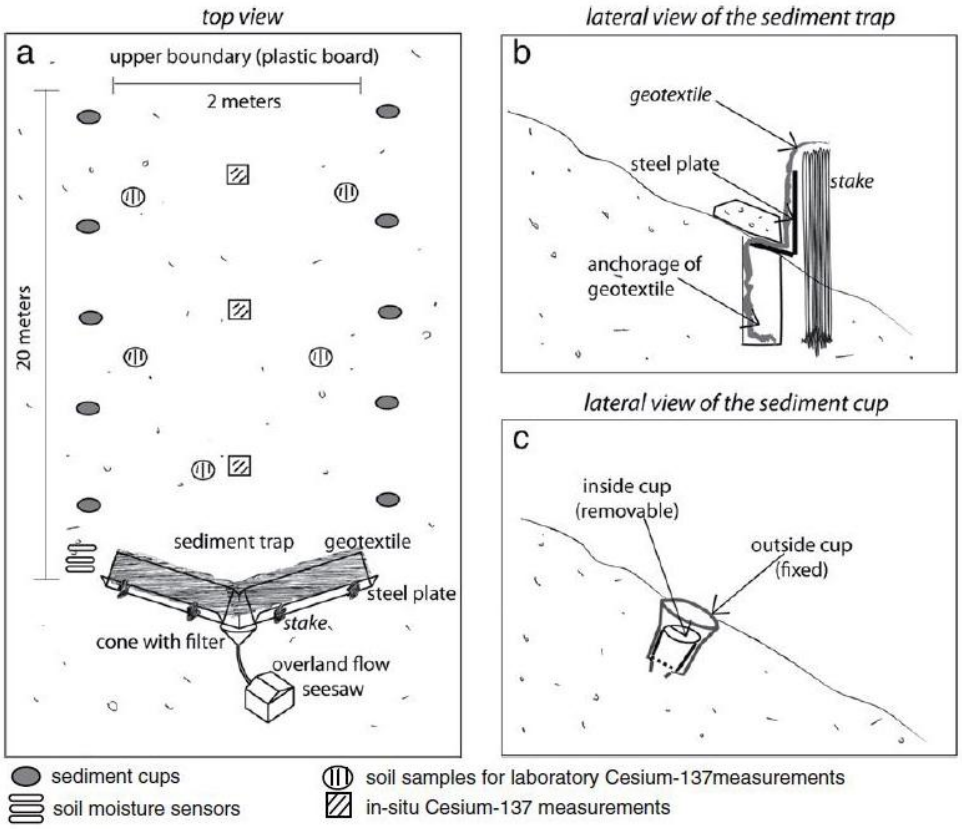
Sprinkling experiments with portable rainfall simulators are an important research tool and are often conducted to assess process dynamics of soil erosion and surface hydrology and to estimate relative differences in erosion susceptibility (as presented above). However, extrapolation of measured sediment yield over time is not possible since the artificial rainfall is not comparable to measured natural rainfall spectra (Iserloh et al. 2013).

A further tool to assess relative differences in erosion activity is sediment cups (Van Dijk et al. 2003). The measurements with sediment cups capture small-scale movement of soil particles that are detached by rain drops as well as animal trampling. The sediment cups do not have fixed boundaries. Thus, the collected data are not quantitative as the size of the area is not defined. However, the method successfully indicated relative differences in erosion activity between different plots and is especially useful to assess small-scale on-site erosion dynamics.

Plot measurements with sediment traps are the most commonly used method of soil erosion measurement. In the presented case study, classical Wischmeier plots with a contribution area of 40 m² (2 m wide × 20 m long) were used (Fig. 27 a). The definition of the contributing area is a critical part when installing sediment traps because artificial side boundaries might cause additional disturbance in the stony soils and potentially channelized surface flow. Consequently, artificial side boundaries were not considered and the plots' locations were chosen to ensure that the topography clearly defined the side boundary. Only the upper boundary was defined, using plastic boards inserted to a depth of 20 cm. The slope sections were separated from the upper slopes by large terraces. The sediment traps (see Robichaud and Brown, (2002) were extended by a V-shaped steel plane below a geotextile to concentrate and measure surface runoff (Fig. 27 b). Soil that was flushed into the geotextile was collected every second week during the growing season from April to November (2007, 2008). The advance installation of sediment traps was important in order to guarantee that soil edges were fully healed and that damaged vegetation had regrown.

Fig. 29 > Composition of all soil erosion measurements with sediment traps and sediment cups and caesium-137 measurements as well as overland flow and soil moisture measurements in the Urseren Valley

A lateral view of sediment traps (b) and sediment cups (c).



Konz et al. 2012

A great advantage of sediment plots is that parameters like precipitation, soil moisture, and surface runoff can be measured parallel to soil erosion. Precipitation was measured with tipping bucket (ECRN-50) rain gauges, (see DecagonDevices 2007), soil moisture with an EC-5 sensor (DecagonDevices), and surface flow with a two-bowl tipping bucket (each bowl having a capacity of 0.5 L). All these devices produced consistent data over the measurement period.

Nonetheless, soil erosion measurement with sediment traps on sub-alpine grasslands is of limited use since erosion events are subject to a high spatial and temporal heterogeneity. It is critical to find a representative location and time period for the measurement with sediment traps and it is not really possible to install the number of repetitions needed to statistically represent an area. A further problem of sediment traps is that they are not suitable for winter measurements because they will be destroyed by snow gliding and avalanches (Fig. 30).

Fig. 30 > Flattened sediment trap due to snow movement after the winter of 2007/2008

Konz et al. 2012

3.1.2 Fallout radionuclides: a tool for assessment of soil erosion rate in alpine areas?

^{137}Cs: a commonly used soil erosion tracer – but is it suitable for alpine areas?

The quantification of soil erosion rates with traditional approaches often requires labour-intensive monitoring programmes. In contrast, fallout radionuclides offer retrospective soil erosion rates because they give a time-averaged estimate of soil movement since fallout commenced in the mid-1950s due to thermonuclear weapon tests and later (1986) the nuclear power plant accident in Chernobyl. In the optimal case, only one field visit may be required to obtain sufficient samples for soil loss estimates to be made (Mabit et al. 2008; Walling and Quine 1990). The strength of the method here is that it incorporates all erosion processes. But the technique is limited to sites where sheet and shallow rill erosion predominates, as deep rilling, gullying, and land sliding involve redistribution of subsoil material having no ^{137}Cs content.

Assessment of soil redistribution rates is commonly based on a comparison of the ^{137}Cs inventory (areal activity density) at individual points in the landscape with that of a "stable" landscape position (also termed a reference site), where neither erosion nor deposition has occurred, assuming a homogenous ^{137}Cs fallout. The Alps were mainly affected by fallout from the Chernobyl accident and hence the ^{137}Cs method can be used here to derive average soil redistribution rates for the period since 1986.

^{137}Cs resampling and stable isotopes: tools for reference site identification?

While the FRN ^{137}Cs has been successfully used to determine sheet erosion rates in arable lowland areas (Mabit et al. 2013), the method was found to produce relatively large uncertainties in Alpine grasslands. Recent research conducted in the Urseren and Piora valleys has demonstrated the potentially high spatial variability of the initial fallout of ^{137}Cs even in undisturbed sites (Polek, 2011; Juretzko 2011). The latter variability was most likely caused by a combination of (i) the partly snow-covered ground in Alpine areas during the Chernobyl fallout event in April 1986, which results in an inhomogeneous ^{137}Cs distribution during snow melt, (ii) uncertainties in finding undisturbed references sites in the highly geomorphologically and anthropogenically active slopes of the Swiss central Alps, and (iii) the general heterogeneous distribution of atmospheric ^{137}Cs Chernobyl fallout. The latter can be addressed by selecting reference sites in the vicinity of the targeted fields that will be subsequently investigated to ensure a similar input of radiocaesium (Mabit et al. 2013). For the Piora Valley, the traditional ^{137}Cs approach could not be applied, as basic assumptions required for the use of this method are not fulfilled (i.e. homogenous deposition of the initial fallout linked to Chernobyl and availability of representative undisturbed sites). Another approach to overcome the uncertainties related to the identification of the reference sites is the ^{137}Cs resampling method. The ^{137}Cs resampling approach allows the erosion rate to re-estimated after resampling a site following a known amount of time. If the ^{137}Cs has changed since the first assessment, the site can be excluded as a reference. This approach has been promoted and used successfully in various regions in Australia (Loughran and Balog 2006; Tiessen et al. 2009; Li et al. 2011). The approach to evaluate the suitability of reference sites based on stable isotope profiles (Meusburger et al. 2013b) will be discussed below.

$^{239+240}$Pu – a more suitable soil erosion tracer for the Alps?

Recently, anthropogenic radioisotopes of plutonium (Pu) have been suggested as new soil redistribution tracers (Schimmack et al. 2002). The two major Pu isotopes (i.e. ^{239}Pu [half-life = 24 110 years] and ^{240}Pu [half-life = 6561 years]) are alpha-emitting actinides which originate from various sources such as nuclear weapons tests, nuclear weapons manufacturing, nuclear fuel reprocessing, and nuclear power plant accidents (Ketterer and Szechenyi 2008). On a global basis, above-ground nuclear weapons testing in the 1950s and 1960s is the main contributor. Pu is contained in the non-volatile fraction of nuclear fuel debris released from reactor accidents such as the Chernobyl accident in 1986. Accordingly, the geographic distribution of Chernobyl Pu fallout is confined regionally to Russia, Ukraine, Belarus, Poland, the Baltic countries, and Scandinavia (Mietelski and Was 1995). Pu deposited from the Chernobyl accident can be distinguished based upon its isotopic composition. The ^{240}Pu/^{239}Pu atom ratio of Northern Hemisphere mid-latitude weapons testing fallout is 0.180 ±0.014 (Kelley et al. 1999), whereas Pu atom ratios of Chernobyl fallout show values of 0.37–0.41 (Muramatsu et al. 2000; Boulyga and Becker 2002; Ketterer et al. 2004).

Soils of the Urseren and Piora valleys have ^{240}Pu/^{239}Pu ratios very close to global, bomb-associated fallout values of 0.18 (Alewell et al. 2014). Eikenberg et al. (2001) also concluded that Pu fallout from Chernobyl was negligible for Switzerland. Like ^{137}Cs, Pu isotopes are strongly absorbed by fine soil particles and their subsequent translocation is mainly related to soil redistribution (Everett et al. 2008; Ketterer et al. 2004, 2011). Only a few studies using Pu as a tracer for soil erosion have been performed so far (Schimmack et al. 2002; Everett et al. 2008; Tims et al. 2010; Hoo et al. 2011, Lal et al. 2013). Since $^{239+240}$Pu is (i) mostly linked to the past nuclear bomb tests, which took place from 1954 to the mid-1960s, and (ii) deposited throughout the year and not connected to a few specific deposition events on snow-covered ground, we found a more homogenous fallout distribution than for ^{137}Cs (Alewell et al. 2014). The coefficient of variance (CV) for reference sites was 29% for ^{137}Cs distribution in the Urseren (n = 6) and 95% in the Piora Valley (n = 7). In contrast, reference $^{239+240}$Pu values had CVs of 10 and 18% for the reference sites at Urseren Valley (n = 6) and

Piora Valley (n = 7), respectively. We conclude that plutonium is a suitable tracer for soil erosion assessment in Alpine grasslands, while the use of ^{137}Cs data is connected to high uncertainties, as a CV of >30% is considered problematic in using fallout radionuclides for soil erosion assessment.

Taken together, $^{239+240}Pu$ at the Urseren and Bedretto/Piora valley sites is bomb-derived with no major impact from the Chernobyl nuclear accident and $^{239+240}Pu$ as a tracer for soil erosion is more homogenously distributed than ^{137}Cs at references sites and thus better suited to assess soil erosion rates in Alpine grasslands. Another advantage of $^{239+240}Pu$ over ^{137}Cs is that $^{239+240}Pu$ activities are measured by quadrupole ICP-MS, which is the preferred tool for routine elemental analysis at thousands of labs worldwide. Both capital and operating costs are reasonable, and sample throughput can exceed 100 samples per day at a cost of less than 50 euros per sample. Furthermore, the long half-life of plutonium ensures long-term availability of the tracer. However, the suitability of the models needed to convert the $^{239+240}Pu$ inventories to soil erosion rates still needs to be evaluated.

3.1.3 Stable isotopes as indicators of soil perturbation

A promising tool to investigate element sources and sinks as well as processes in ecosystem biogeochemistry is isotope tracers (for an overview, see Kendall & McDonnell 1998). Especially, a stable carbon (C) isotope signature can provide useful information about soil redistribution (Fox and Papanicolaou 2007; Turnbull et al. 2008). Two approaches to indicate soil disturbance, with no transition from C3 to C4 vegetation, were tested for their suitability in Alpine areas. The first one aims to track soil erosion in hill slope transects from uplands (erosion source, oxic soils) to adjacent wetlands (erosion sink, anoxic soils), as they often occur in Alpine environments. The second aims at indicating long-term disturbance of oxic upland soils by decreasing correlation of $\delta^{13}C$ versus soil organic carbon (SOC), $\delta^{15}N$ versus N content, and $\delta^{15}N$ versus C/N ratio, respectively.

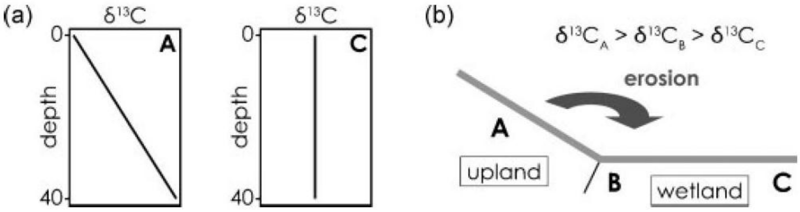
Soil depth profiles of the ^{13}C stable isotope

The depth distribution of stable carbon isotopes (^{12}C and ^{13}C) reflects the combined effects of plant fractionation processes and microbial decomposition (Krull and Retallack 2000). In undisturbed oxic mineral soils, we can generally assume a gradual enrichment of $\delta^{13}C$ and $\delta^{15}N$ with depth parallel to the decrease in C and N content due to enhanced degradation of soil organic matter (Nadelhoffer and Fry 1988; Schaub and Alewell 2009; Alewell et al. 2011). Theoretically, the Suess effect (the influence from fossil fuel burning on stable carbon isotope signature) has to be considered. However, studies by Alewell et al. (2011) and Schaub and Alewell (2009) pointed to a negligible Suess effect on the stable isotope signature of bulk soil material.

Tracking of sediment transport to wetlands using stable carbon isotopes

Different carbon isotope signatures can be expected for uplands and adjacent wetlands soils (Schaub and Alewell 2009). In the aerobic environment of upland soils, oxidative processes dominate during decomposition of plant material. Isotopic fractionation during these processes leads to an enrichment in the heavier carbon isotope of the residues (^{13}C) as the lighter ^{12}C will preferentially be involved in chemical reactions (Fig. 31 a). In contrast, the anoxic conditions of wetland soils result in incomplete decomposition of organic material by anaerobic bacteria. Carbon compounds are preserved and keep their original (plant) isotopic signature (Fig. 31 b). Therefore, $\delta^{13}C$ of SOC in wetland soils can be assumed to be lighter than that of upland soils.

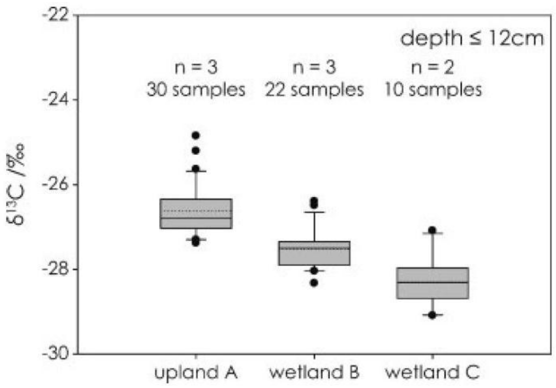
Fig. 31 > Theoretical $\delta^{3}C$ depth profiles in uplands (A) and undisturbed wetlands (C), and (B) theoretically expected influence of erosion on the $\delta^{3}C$ of wetlands disturbed by erosion (B)



Schaub and Alewell 2009

The carbon isotope signatures of the upper soil horizons of wetland soils influenced by erosion have intermediate $\delta^{13}C$ values of –27.5 ±0.5%, that is, between the $\delta^{13}C$ of upland soils with values of –26.6 ±0.6% and the reference wetland with values of –28.3 ±0.6% (Fig. 31 b). There is a significant difference between the reference wetlands and the wetlands influenced by erosion ($p < 0.01$). The stable carbon isotope signatures of lower horizons (12–24 cm) correspond in both wetland types (data not shown). The approach could also be successfully applied in the Piora Valley (Brun 2012).

Fig. 32 > Delta-^{13}C for two depth steps (0–12 cm and 12–24 cm). Boxplots indicate the median (straight line), mean (dotted line), 10th, 25th, 75th, and 90th percentiles, and outliers (dots)



Schaub and Alewell 2009

Tracking of sediment transport to wetlands using stable oxygen isotopes

Stable oxygen isotope signatures were also explored as indicators for soil erosion in transects from upland to wetland soils. The stable oxygen isotope signature ($\delta^{18}O$) of soil is expected to be the result of a mixture of components. The $\delta^{18}O$ of soils should provide information about the relative contribution of organic matter versus minerals. As there is no standard method available for measuring soil $\delta^{18}O$, a measurement method for single components using a high-temperature conversion elemental analyser (TC/EA) was developed (Schaub et al. 2009). The $\delta^{18}O$ was measured in standard materials (IAEA 601, IAEA 602, Merck cellulose) and soils (organic and mineral soils) to determine a suitable pyrolysis temperature for soil analysis. The test of several substances within the temperature range of 1075 to 1375 °C indicated a suitable pyrolysis temperature of 1325 °C (Schaub et al. 2009).

For the Urseren Valley, a sequence of increasing $\delta^{18}O$ signatures from phyllosilicates to upland soils, wetland soils, and vegetation was found (Schaub et al. 2009). Thereby uplands and wetlands not affected by erosion significantly differed in $\delta^{18}O$ values ($p < 0.05$; Fig. 33). The upper horizons (0–10 cm) of upland soils have a mean $\delta^{18}O$ between 5 and 15% while $\delta^{18}O$ signatures of reference wetland soils vary between 15 and 20%. The $\delta^{18}O$ values of wetland soils are rather similar to those of the vegetation cover, which reflects the high proportion of organic matter. The $\delta^{18}O$ signature of upland soils is less positive, indicating the influence of mineral components. No indication of major isotopic fractionation during the decomposition of organic matter was found. As such the $\delta^{18}O$ value of soils seems to be the result of pure mixing of components with differing $\delta^{18}O$ signatures.

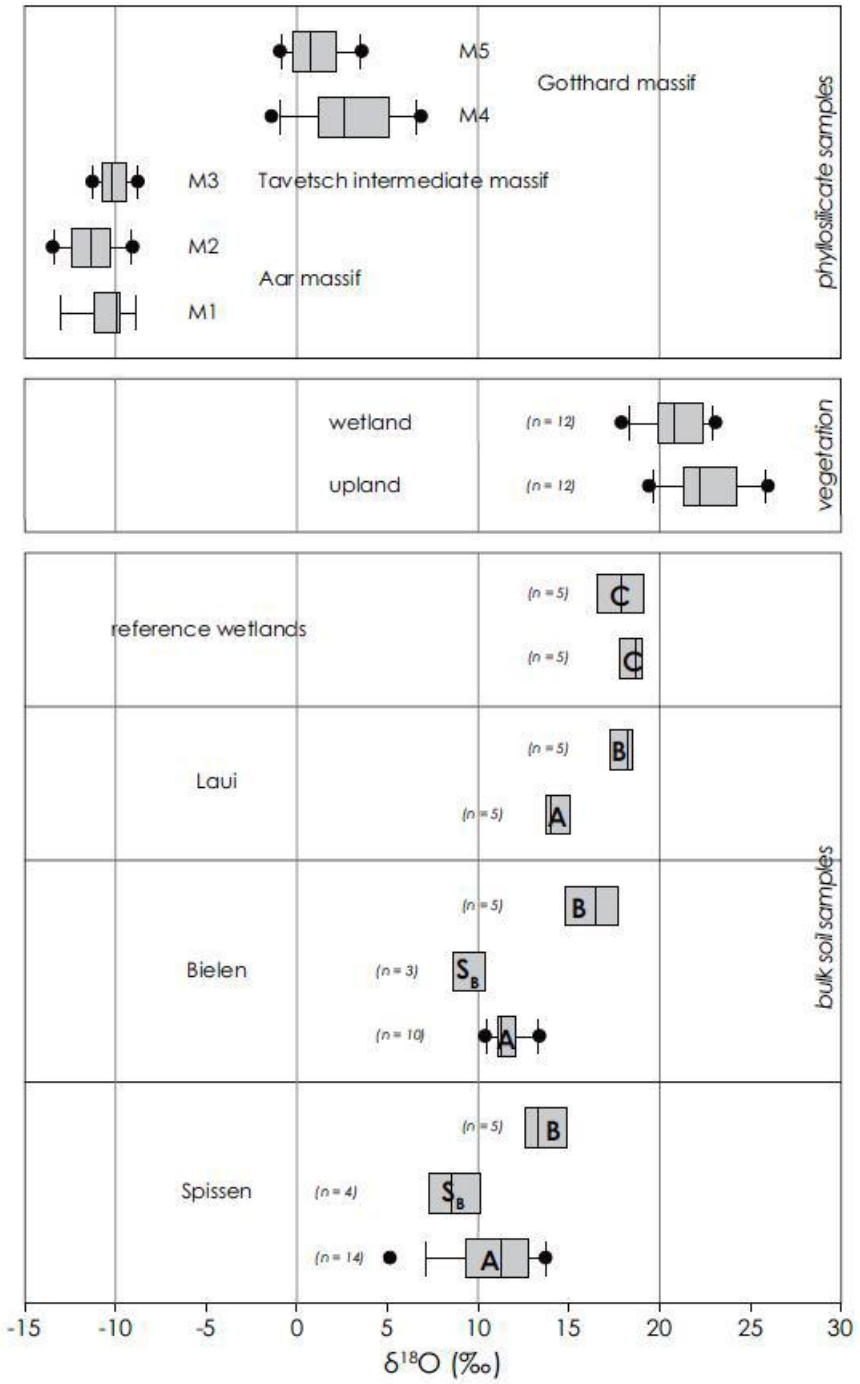
The latter can be tested by calculating the mixing ratio for organic soil with fractions of 80% organic matter ($\delta^{18}O = 21.0 \pm 1.5\%$) and 20% mineral components (mean $\delta^{18}O$ of phyllosilicates from the Gotthard massif = 1.8 ±2.0%). The resulting calculated mixing signature for wetland soils is 17.2 ±1.6%, which is in the range of measured $\delta^{18}O$ signatures for reference wetlands.

All three investigated transects were characterized by similar carbon contents. However, only two of the sites have similar $\delta^{18}O$ values, with a mean of around 12% (Spissen and Bielen). At the third site (Laui) more positive values (Fig. 33) were observed. Possibly the soil of this site developed from another substrate with a different $\delta^{18}O$ signature (e.g. moraine material).

The prerequisite (for the use of $\delta^{18}O$ as a tracer for soil erosion) of different isotopic signatures of upland and reference wetland soils (assigned as C) was fulfilled for soils in the Urseren Valley (Fig. 33). Intermediate $\delta^{18}O$ values for wetland soils adjacent to an upland can consequently be interpreted as mixing of soil erosion material with the organic wetland soil (Fig. 33). A significant ($P < 0.05$) shift towards upland soil signatures was determined for the two sites, Spissen and Bielen, where erosion is also visually documented as a sediment layer covering parts of the wetland (S_B, Fig. 33). Consequently the application of $\delta^{18}O$ is promising but restricted to sites with similar soil bedrock material.

Fig. 33 > Stable oxygen isotope signature of phyllosilicates of central Switzerland

repeated measurement of samples from the same rock sample, aboveground vegetation from upland and wetland sites in the Urseren Valley, and soil samples from sites in the Urseren Valley (A = upland, B = wetland with erosion influence, S_B = sandy layer at position B, C = reference wetland unaffected by erosion).



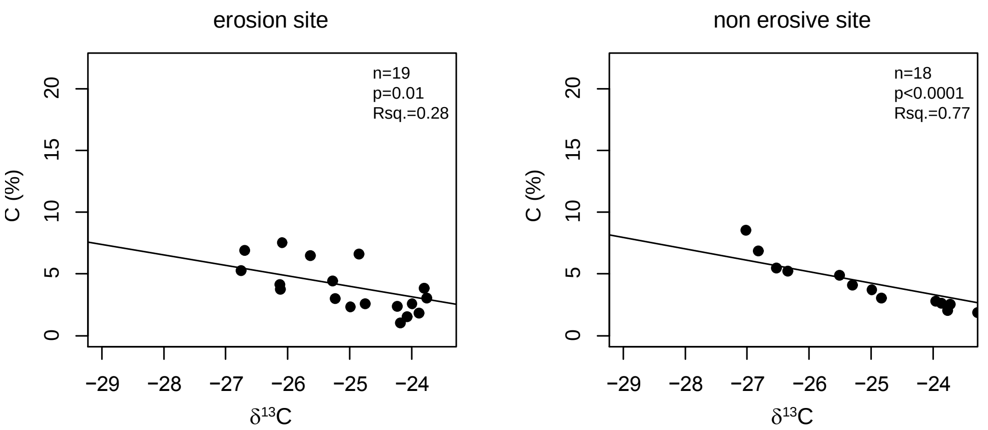
Schaub et al. 2009

Long-term soil disturbance indicated by $\delta^{13}C$

A key component for understanding soil degradation is SOC. SOC complexes stabilize soil structure and increase the reactive surface of the soils (Gregorich et al. 1994; Lal 2004). During soil erosion, however, SOC is the first component to go into suspension in the form of small colloidal particles and is preferentially exported (Bilgo et al. 2006; Polyakov and Lal 2004; Watung et al. 1996), and mineralization and leaching of SOC may be fostered (Lal 2003). The latter processes result in a shift of $\delta^{13}C$ values and thus a deviation from the correlation between the $\delta^{13}C$ and C contents of oxic soils. In an undisturbed soil $\delta^{13}C$ is strongly linked to soil carbon content since SOC decomposition is accompanied by isotopic fractionation. Parallel to the depth increase of $\delta^{13}C$ in undisturbed oxic soils, we usually find a decrease in SOC content.

The correlation coefficients (r) between $\delta^{13}C$ and SOC were found to be over 0.80 for all uneroded upland soils. However, a deviation from the strong correlation between $\delta^{13}C$ and SOC of upland soils is paired with visual erosion damage at the sites (Fig. 34). Slope transects with no visible erosion output have >0.80 r (Laui A = –0.97, Bielen AA= –0.93, Spissen AAA = –0.89) whereas sites prone to erosion have values equal to and smaller than 0.80 r (Bielen A = -0.78, Spissen AA = -0.80, Spissen A = -0.57).

Fig. 34 > Carbon content vs. $\delta^{3}C$ for eroded and uneroded sites in the Urseren Valley



data after Schaub and Alewell 2009

The presented approach could be validated for an eroded site in Korea (Meusburger et al. 2013b).

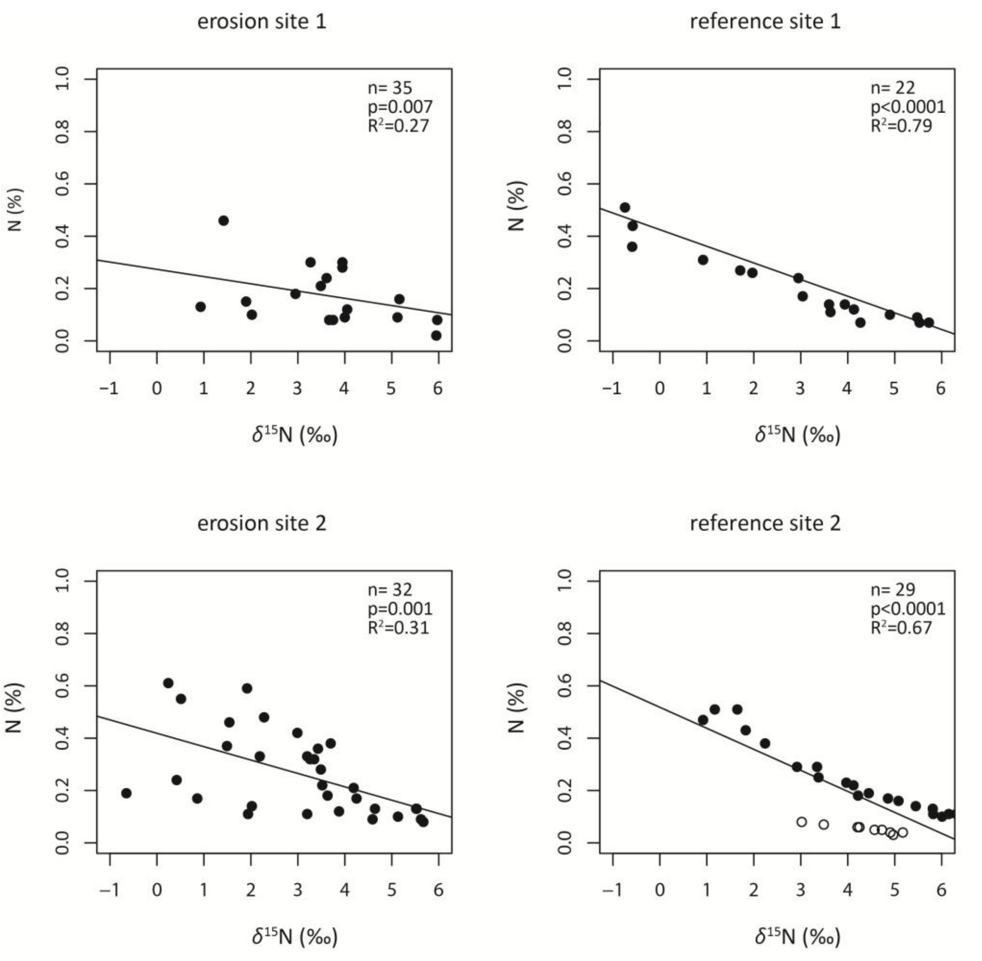
Long-term soil disturbance indicated by $\delta^{15}N$

The concept of correlation was further applied to assess the suitability of $\delta^{15}N$ as a soil erosion tracer. $\delta^{15}N$ is enriched with soil depth due to an increasing proportion of microbially altered N transported downward in the soil profile and also aerobic mineralization (Mariotti et al. 1980).

However, the investigated alpine grasslands sites are affected by farmyard manure. The isotopic composition of animal manure resulted in a mean $\delta^{15}N$ value of 5.8 $\pm$1.1 (Alewell et al. 2008) and is significantly higher than for unmanured sites, which have a mean of 0.18 $\pm$0.8, and in the same range as the manured soils, which have a mean of 5.2 $\pm$0.7 (Schaub 2008). Consequently, all grassland soils, either oxic or wetland, exhibited substantially higher δ^{15}N values. The δ^{13}C signal is not influenced by the application of manure since it is in the same range as that of the vegetation (Schaub et al. 2009).

Because of the strong influence of manure in the investigated alpine sites, the approach was tested in a remote South Korean mountain site. For these soils, plotting total nitrogen content (%) versus δ^{15}N resulted in clearly higher correlations with $r^2 = 0.79$ and $r^2 = 0.67$ for undisturbed sites than for erosive sites ($r^2 = 0.27$ and 0.31, respectively, Fig. 35). At the undisturbed site 2, some samples plot slightly below the trend line (hollow dots). These are samples below a soil depth of 21 cm with a higher sand content, resulting in a dilution of the nitrogen content. In the case of unfertilized soils, the correlation between δ^{15}N and nitrogen content for the soil profile appears to be a suitable soil disturbance indicator.

Fig. 35 > Total nitrogen content (%) vs. δ^{5}N for reference sites and erosion transects. Sampling points of one transect are grouped in the same chart



Delta-^{15}N versus C/N ratio as a soil disturbance indicator

Delta-^{15}N is further functionally related to the ratio of organic C to total N in soil (C:N ratio). Soils with a small C:N ratio are more likely to be enriched in ^{15}N, as ^{14}N is preferentially lost during the decomposition of organic matter and transformations of N species (Mariotti et al. 1980; Robinson 2001; Conen et al. 2008). In a recent study in New Zealand, it was shown that the C:N ratio explains a large proportion of the observed variation in $\delta^{15}N$ values for all major land uses (Stevenson et al. 2010). Thus if we assume that $\delta^{15}N$ values cover a relatively narrow range at any particular C:N ratio in soils, then any substantial loss or gain of N should mostly result in the loss or gain of ^{15}N-depleted forms. The latter would result in larger or smaller $\delta^{15}N$ values than usual at the observed C:N ratio and could serve as a soil disturbance indicator.

A correlation was found between $\delta^{15}N$ values and C:N ratios in different (semi)-natural northern Eurasian soils (Conen et al. 2013), which is congruent with the data presented for New Zealand. However, soil samples with evidence of disturbance to the N cycle mostly fall outside the dotted lines (26 of 36 data points). The available data also indicate that soil disturbance that accelerates the loss of ^{15}N-depleted forms (fire or overgrazing) results in larger $\delta^{15}N$ values while soil disturbance that stimulates the accumulation of organic matter (afforestation) results in smaller $\delta^{15}N$ values than expected at a particular C:N ratio. This approach still needs to be tested for alpine grassland sites.

To conclude, stable isotopes are valuable qualitative indicators for soil disturbance if the underlying assumptions are considered.

3.2 Mapping erosion features and erosion risk factors on catchment scale

Mapping of shallow landslides with aerial photographs

Field surveying and mapping of erosion features in the Alps is impeded by site inaccessibility. Even in remote areas, remote sensing techniques allow the surveying of erosion features such as landslides, gullies, or vegetation disturbance for the collection of input data for soil erosion models (De Jong 1994; Vrieling 2006). To assess the shallow landslide development over time for the three valley sites, we used aerial photograph interpretation. A geographic information system (ArcDesktop 9.1, ESRI) was used to collect, superimpose, and analyse the spatial data layers. Landslides were mapped by visually vectorizing the affected area from aerial photographs. Photographs were selected from a series of aerial photographs that are available at Swisstopo at a scale of at least 1:12 000. In order to allow for the local comparison of the individual landslides between different years the photographs were georeferenced and orthorectified using the ENVI software package (Version 4.0) with the help of ground control points, the DEM, and the camera calibration protocol supplied by Swisstopo. For sites with high relief, it is recommended that each slope be georeferenced separately.

At some locations, difficulties in distinguishing gravel fields from landslides may occur. Thus, field verification is necessary to check the inventory map produced by aerial photograph interpretation. During these field surveys in the Urseren Valley, eight landslides were investigated in more detail to assess the accuracy of the GIS mapping method. The areas of those eight landslides determined during the field survey were compared with the corresponding areas achieved by photo interpretation. The latter resulted in an error of ±10% (Meusburger and Alewell 2008).

The aerial photograph mapping is a very useful tool and probably the only one available to establish shallow landslide inventory maps for different years. Today new satellites offer a comparable spatial resolution, but they do not reach back into the past. A disadvantage is that the method is very labour-intensive and also subjective; however, recently a new automatic landslide mapping method was developed that has the potential to resolve this problem (Wiegand et al. 2013). Still the performance of aerial photographs for automatic land cover or landslide classification is limited due to the low number of spectral channels.

Land-cover mapping through satellite image classification

In high relief regions with rugged topography, soil erosion risk factors are needed on a detailed scale. Jetten et al. (2003) even proposed that more benefit for soil erosion assessment might be gained by improving spatial information for model input and validation than by adapting models to a specific landscape. Satellite imagery can provide such valuable spatial information mainly on vegetation parameters to improve the performance of soil erosion assessment (De Jong 1994; De Jong et al. 1999; Tweddales et al. 2000; Jain et al. 2002; De Asis and Omasa 2007; De Asis et al. 2008).

Compared to aerial photographs, issues of land cover classification and estimation of vegetation abundance are feasible with satellite images, because they offer more spectral bands. In this case study, QuickBird data were used since they offer a very high spatial resolution, which is needed due to the heterogeneity of alpine areas.

In arable regions vegetation cover is characterized by a strong temporal dynamic (Vrieling 2006). This dynamic is less pronounced in alpine areas with permanent grassland because disturbances in vegetation cover due to trampling, overgrazing, cutting, snow transport, and landslides recover slowly and do not follow a seasonal dynamic as on arable land, where ploughing and harvesting are applied. The QuickBird image was taken in autumn. This is a relevant time since bare soil areas that have not recovered during the growing season are persistent and are prone to winter damages and subsequent snowmelt, when the highest erosion rates can be expected.

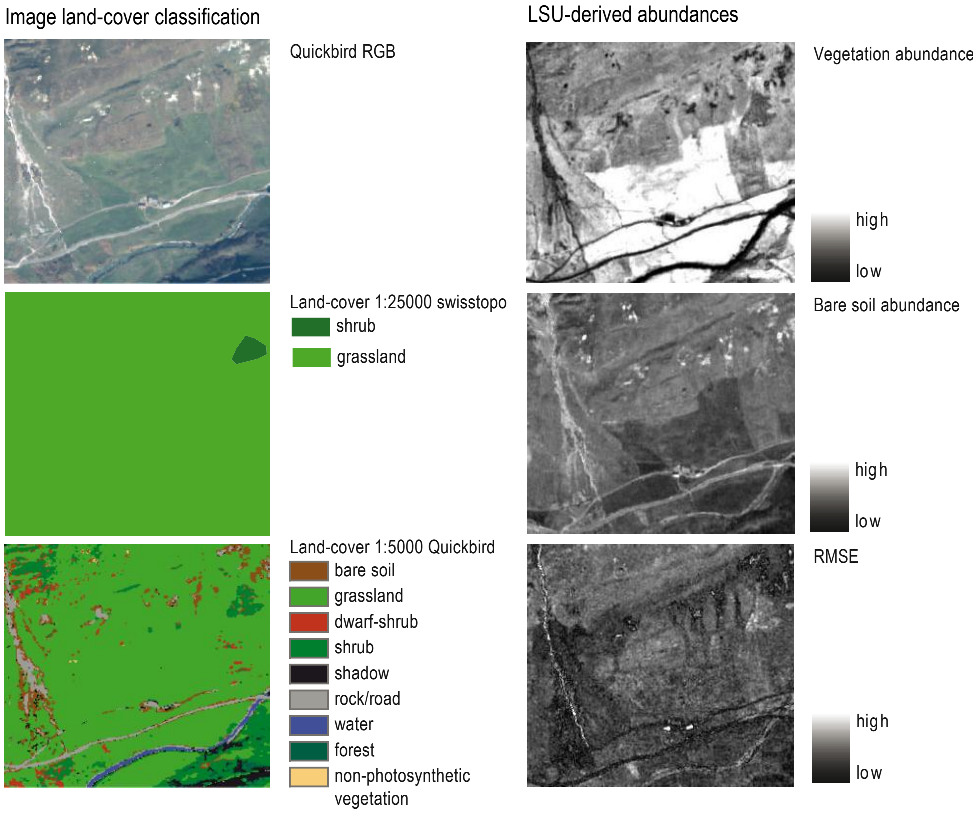
Land-cover mapping was done using supervised classification based on a maximum likelihood classifier in the ENVI 4.3 software (Research Systems Inc., Boulder, CO). Land cover was classified into the following nine categories: forest, shrub, dwarf shrub, grassland, non-photosynthetic vegetation, snow, water, bare soil, rock, and the artificial category shadow, which was introduced because of the distinct topography of the study area. For higher elevations, especially in autumn, the problem that spectral signatures of bare soils and non-photosynthetic vegetation become similar occurs. Thus, subsequent to the supervised classification, all pixels classified as bare soil above 2000 m a.s.l. with an NDVI greater than 0.2 were changed to non-photosynthetic vegetation. This threshold and the elevation above 2000 m a.s.l. were chosen because they produced the best classification results compared to the ground truth data.

The overall accuracy of land-cover classification was 93.3% (3000/3217 pixels), which is very good. In total, only 0.81% of the dataset remained unclassified. The land-cover classes of bare soil, grassland, rock, snow, water, forest, and shadow have high classification accuracies (82–100%), while the dwarf shrub, shrub, and non-photosynthetic vegetation classes have lower classification accuracies (67–78%). However, the percentage values of each land-cover class within the study site could not be determined due to large shadowed areas. It was possible to derive more details with the satellite-based classification in comparison to the Swisstopo VECTOR25 dataset (Fig. 36, left).

Particularly, the detection of bare soil areas that have the highest potential erosion rates is of great interest. Further, it was possible to map dwarf-shrub vegetation that behaves differently with respect to several erosion-rated processes (Meusburger et al. 2013a).

Fig. 36 > Sub-image of the QuickBird scene showing (left) the resulting land-cover map compared to land-cover information of the Swisstopo 1:25 000 Vector dataset and (right) normalized abundances of vegetation, bare soil, and RMSE for the linear mixture model

The lighter the colour, the higher the proportion within the pixel.



Meusburger et al. 2010a

Mapping of vegetation cover abundances with satellite data – the method

As discussed above, the intact vegetation cover is the most important factor preventing soil loss by water erosion in alpine areas. Land-cover classification through image classification is one of the most commonly used techniques to assist erosion assessment. Besides image classification, spectral indices and spectral unmixing are used, which are especially useful to derive vegetation abundances (Vrieling 2006). Spectral indices, for example the Normalized Difference Vegetation Index (NDVI), have been used for the direct mapping of vegetation cover (Liu et al. 2004) or to improve the mapping of the RUSLE C-factor (De Jong 1994; De Jong et al. 1999). For some regions, however, vegetation indices were found to have low correlation with the C-factor (De Jong 1994; Tweddales et al. 2000) due to the sensitivity of the NDVI to vitality of vegetation (De Jong 1994). With respect to C-factor mapping, this is problematic since dry vegetation can also efficiently protect the soil surface from water-induced erosion. The problem was overcome by using linear spectral unmixing (LSU). LSU is based upon the concept that a signal detected by a sensor of a single pixel is frequently a combination of numerous disparate signals, for example, in our case, of soil and vegetation. The knowledge of the spectra of the pure components (soil and vegetation) allows the signal of the mixed pixel to be disaggregated into its individual percentage components. Asner and Heidebrecht (2000) were able to successfully map non-photosynthetic vegetation using the LSU technique.

Mostly, LSU has been used with satellite systems with a medium spatial resolution and medium to high spectral resolution (higher number of spectral bands). However, the alpine environment exhibits a spatially complex and heterogeneous biogeophysical structure, where abundances of bare soil, vegetation, and rock vary at small scales. The problem of spatial heterogeneity could be overcome by the use of very high-resolution satellite imagery offered by the QuickBird (pixel resolution = 2.4 m) satellite.

Mapping of vegetation cover abundances with satellite data – results

To evaluate the performance of the satellite-based mapping, ground truth data are needed. Therefore the fractional vegetation cover (FVC) was assessed at several spots in the valley within two weeks after the image was taken. FVC in the field is defined here as the percentage of an area covered by grass. The inverse of FVC equals the fraction of bare soil.

The above discussed land-cover map was applied prior to the LSU analysis in order to stratify the image. The resulting regression between vegetation abundance derived from ground truth FVC and NDVI as well as mixture-tuned matched filtering, which is a second unmixing technique, was significant (r^2 = 0.64, r^2 = 0.71, respectively). But the best results were achieved for LSU (r^2 = 0.85). The abundance images derived using LSU for vegetation, bare soil, and RMSE are shown in Fig. 36 (right). The maps reveal high spatial variability that represents the different cover conditions in the study area. Vegetation abundance is highest close to the valley. The evident bare soil areas correspond to landslides, overgrazed areas, roads, and river banks. The RMSE image shows no recognizable pattern for the grassland areas, indicating that the chosen end members are valid.

In contrast, the estimation of the bare soil abundance (r^2 = 0.39 for LSU) yielded worse results, most likely due to the high spectral variability of bare soil at the study site and the low spectral resolution of the QuickBird imagery.

The successful mapping of vegetation abundances may be due to the very distinct reflectance curves of vegetation if they are compared with the reflectance of bare soil, rock, and water. The vegetation abundance maps cannot be used directly as model input due to strong deviations from the 1:1 line. But the regression equation established between the vegetation abundance maps and the ground truth data can be used to convert the vegetation abundance maps to the FVC map.

In general, QuickBird is not the “optimal” sensor for unmixing approaches because of the small number of spectral bands. Especially, for the mapping of bare soil abundances, the system seems to meet its limits. One main reason is the lack of shortwave infrared channels, which are important for mineral and rock discrimination since minerals have numerous absorption bands in this wavelength range.

The problem of mapping bare soil abundance might be solved by using either images of the WorldView-2 satellite (launched in October 2009), which provides eight spectral bands at 1.8 m, or sufficient ground truth data.

Benefit of high-resolution input data for soil erosion risk assessment

Detailed mapping of land cover and separation of different vegetation types is crucial for soil erosion assessment because different vegetation types produce different sediment yields (Isselin-Nondedeu and Bedecarrats 2007). Consequently the produced abundance maps of vegetation are preferable to only parameterizing thematic land-cover classes because they retain spatial variability. The RUSLE and PESERA models were used to illustrate the decisive effect of the produced land cover and vegetation abundance map (Meusburger et al. 2010b). Using the former low-resolution dataset with 100% FVC, both models underestimate erosion rates by almost an order of magnitude. Assuming 100% vegetation cover, PESERA estimates approximately zero soil erosion. By introducing the LSU-derived FVC, the modelled rates of both models approximate the modelled values based on the field-measured input data (Meusburger et al. 2010b). For PESERA it is noteworthy that even 0% FVC produces very low erosion values considering the high slope angle and erosivity.

3.3 Evaluation of suitable models for soil erosion risk mapping

Overview of different soil erosion models

Erosion models can be classified into four categories: empirical, conceptual, physically based, and stochastic. The empirical models are based on observations and statistical relationships arising from the analysis of a large number of experimental data. They are the simplest type of model, and thus they are often criticized because they tend to ignore the heterogeneity and non-linear behaviour of erosion processes (Kinnell 2005). However, they are the most commonly used models at the regional scale (Merritt et al. 2003).

The most important empirical models are the USLE (Wischmeier and Smith 1978) and its revised version, RUSLE. These two models calculate soil loss (A) as a product of the six main erosion factors, which are rainfall erosivity (R), soil erodibility (K), topography (slope steepness and length, LS), land use and management (C), and a support practice factor (P). The data basis for the RUSLE consists of more than 10 000 measurement years and thanks to its undoubted points of strength, some parts of this model have been incorporated in other models, such as Unit Stream Power Erosion Deposition (USPED), CREAMS, AGNPS, SWAT, and ANSWER (Merritt et al. 2003).

Conceptual models lie somewhere between physically based and empirical models because they are based on physical laws and mathematical equations such as, for example, the continuity equation for water and sediments, and empirical relations.

Physically based models try to describe the erosion process through the fundamental physical equations describing stream flow and sediment production in the catchment (mass balance and conservation, nutrient balance and sediment yield). This kind of model considers the complex interactions among various factors and their variability in space and time. Physically based models have the advantage that they can be developed further while deepening the knowledge of natural processes. The model input parameters are measurable, but the number of parameters needed is often too high. In particular, parameterizing the heterogeneous behaviour of soil is often an impossible task, especially for larger areas. As a consequence, calibration and approximation are required, which creates uncertainties and limits the transferability to other regions. Thus, physically based models are more useful at plot scale. At catchment or regional scale, they lose significance. The LImburg Soil Erosion Model (LISEM) and WEPP belong to this model category, among which WEPP is the most frequently applied physically based model (Flanagan and Nearing 1995). WEPP analyses the different processes, both hydrologic and erosive, using a variable time scale. A main limitation of this model is that it works on a maximum area of a few hundred hectares. At European scale, the physically based model EUROSEM, developed on the basis of WEPP and PESERA, has been implemented (Morgan et al. 1998; Kirkby et al. 2008).

Erosion models already applied in mountain regions

Alpine areas have a high potential soil erosion risk associated with the extreme climatic and topographic conditions. The magnitude of actual soil erosion rates is, however, highly uncertain due to the spatial heterogeneity of erosion risk factors (e.g. vegetation and snow movement), which causes difficulties in extrapolating sediment measurements on plot scale to larger regions (Helming et al. 2005). Especially the implementation of snow movement will be a future challenge. For arable lowlands, several models have been developed for soil loss quantification (e.g. USLE, RUSLE, LISEM, WEPP, PESERA, and USPED). The validity of these models has to be carefully considered for alpine regions (Van Rompaey et al. 2003a; Van Rompaey et al. 2003b). So far, for the entire European Alps, only the USLE and PESERA models have been used (Joint Research Center Ispra 2009b, a).

The most commonly used soil erosion models in mountain areas are as follows:

> The USLE and its revised version RUSLE are simple and easy-to-use models and have mostly been applied for large-scale assessments (Friedli 2006; Joint Research Center Ispra 2009a)
> USPED can be an alternative to the RUSLE model, because it considers both erosion and deposition. The second main difference from RUSLE is that USPED is a transport-limited model (it considers erosion as limited by the ability of the channel to drain the eroded material), while RUSLE is erosion-limited (the soil loss is limited by the presence of detachable material). The first can be applied after the removal of cells close to channels, which cause a large overestimation of soil loss (Cencetti et al. 2005; Warren 1998).
> WEPP was developed based on data of 1600 plot years of natural runoff, including most of the data used to develop the USLE (Tiwari et al. 2000). Evans and Brazier (2005) showed an overestimation of soil for the WEPP model for both grasslands and cultivated areas. Our studies disagree with these results since soil loss measured with sediment traps was underestimated by WEPP for the period 2007–2008. Other problems derive from the overestimation of the effects of climate and topography (Evans and Brazier 2005), while the land cover effect is underestimated (Meusburger et al. 2010b).
> PESERA (Kirkby et al. 2003) is a physically based model but has a low input data requirement. A runoff threshold is applied, which depends on soil properties, the surface, and the vegetation cover. The runoff threshold is reached when water can no longer infiltrate the soil. The model calculates sheet and rill erosion rates at regional scale. When applied to alpine slopes (Meusburger et al. 2010b), it did not

show significant correlations with measured soil erosion values. One problem is due to the fact that (i) mass transfer from cell to cell on a mountain slope is not considered and (ii) the model is not sensitive enough to changes in fractional vegetation cover.

Generally it can be stated that advanced models such as WEPP are less frequently used in mountain environments than empirical ones because of their complexity and large number of input data. Despite all its simplifications, the USLE/RUSLE model has the advantage of great flexibility: even though it was developed to measure soil loss in cultivated areas, many equations have been advanced in order to adapt the factors to different environments.

At regional and national scales, the accuracy of input data rapidly decreases. Since a balance between model complexity and data demand is desirable, the USLE seems to be the appropriate choice. In addition, erosion processes change with scale, and models such as the USLE cannot capture this. Thus, USLE may be used rather conceptually to obtain a relative ranking of soil loss risk.

4 > Future development and likely scenarios for soil stability in the Alps

4.1 Possible effects of climate variability

Precipitation and sheet erosion – trends in rainfall erosivity

Since the capacity of air to hold moisture is a function of temperature, global warming is likely to increase the frequency and magnitude of heavy precipitation events (Fowler and Hennessy 1995). These changes in interannual climate variability and weather extremes may be the key impacts of climate change (IPCC 2007; Borgatti and Soldati 2010). A good proxy for an extreme rainfall event that is relevant for soil erosion is the RUSLE R-factor. For Switzerland, the RUSLE R-factor showed a significant increasing trend (Mann-Kendall trend test for single months over the 22 investigated years) of rainfall erosivity for the months May to October and a significant decreasing trend for February. This increasing trend in the R-factor corresponds to climate change scenarios that predict an increase in high intensity precipitation events for many parts of Central Europe and the Alps (Christensen and Christensen 2003; Schär et al. 2004; Frei et al. 2006). Further, the trends are supported by an assessment of thunderstorms distribution for Europe, where the highest frequency was observed in the vicinity of the Alps for the season April to October (van Delden 2001). A decreasing erosivity trend was identified for February. Winter precipitation is predicted to increase by 10% for Switzerland (OcCC/ProClim 2007) and winter minimum temperature is expected to rise (Beniston 2003). The combined effect of these developments is expected to increase soil erosion by runoff, but will not necessarily result in an increase of high intensity events as defined by the (R)USLE R-factor.

The trends are congruent with the findings of Mueller and Pfister (2011). Their rainfall times series with a high temporal resolution of 1 minute for the Emscher-Lippe catchment (central western Germany) showed for all investigated stations an increase of rainstorm events with erosion-relevant rainfall intensities; the trend was more pronounced over the last 35 years. This increase of rainstorm events over the last 35 years was more pronounced in the summer season (July–September), but increases were also detected in the other seasons. The high intensity storm events occurred only about 4–15 times yr^{-1}; the estimated trend increases of up to 0.5 events yr^{-1} could therefore result in a multiplication of the occurrence of these storm events within only a few decades.

For the alpine region of Tyrol, De Toffol et al. (2009) analysed storm events and found no significant trends for time series of short durations (15–60 minutes for the period 1950–2000); however they detected an increase in the number of extreme events of shorter durations.

Soil erosion is highly sensitive to the intensity of individual precipitation events (Nearing et al. 2004). However, changes in rainfall erosivity cannot be directly linked to soil erosion risk due to the seasonal variability of vegetation cover. However, three out of the six months with significant increasing rainfall erosivity trends can also be expected to have relatively sparse or instable vegetation cover. Most cultivated areas (e.g. winter crops, corn, vegetable fields) as well as alpine grasslands will have low vegetation cover in May (Vandaele and Poesen 1995; Le Bissonnais et al. 2002; Konz et al. 2009) and will be harvested or grazed in September/October (Favis-Mortlock and Boardman 1995; Leek and Olsen 2000; Helming et al. 2005). Thus they will be susceptible to erosion during snowmelt and/or heavy rain events. Consequently the added effect of high rainfall erosivity in May, September and October will most likely result in an increased soil erosion risk for Switzerland (Meusburger et al. 2012). Several studies have shown that air temperature in the alpine region is changing more rapidly than the global average (Beniston 2006; Hari et al. 2006). This results in another impact of climate change which is associated with a shift from snowfall to rainfall (Fuhrer et al. 2006). The decrease in the number of snowfall days might translate to increases in days of rainfall and subsequently erosion by storm runoff is liable to increase (Fuhrer et al. 2006). The snow cover in winter may protect the soil from freezing, which is important for snowmelt infiltration and runoff generation (Stahli et al. 2001; Bayard et al. 2005). Consequently, higher levels of rainfall and changes in freezing/thawing cycles can also be expected to increase soil erosion due to the sparse vegetation cover or lack of it in formerly snow covered areas in winter and early spring. Also, changes in evapotranspiration and in soil surface conditions, such as soil moisture, surface roughness sealing and crusting, may change with shifts in climate, hence impacting erosion rates (Fuhrer et al. 2006).

Precipitation and landslides

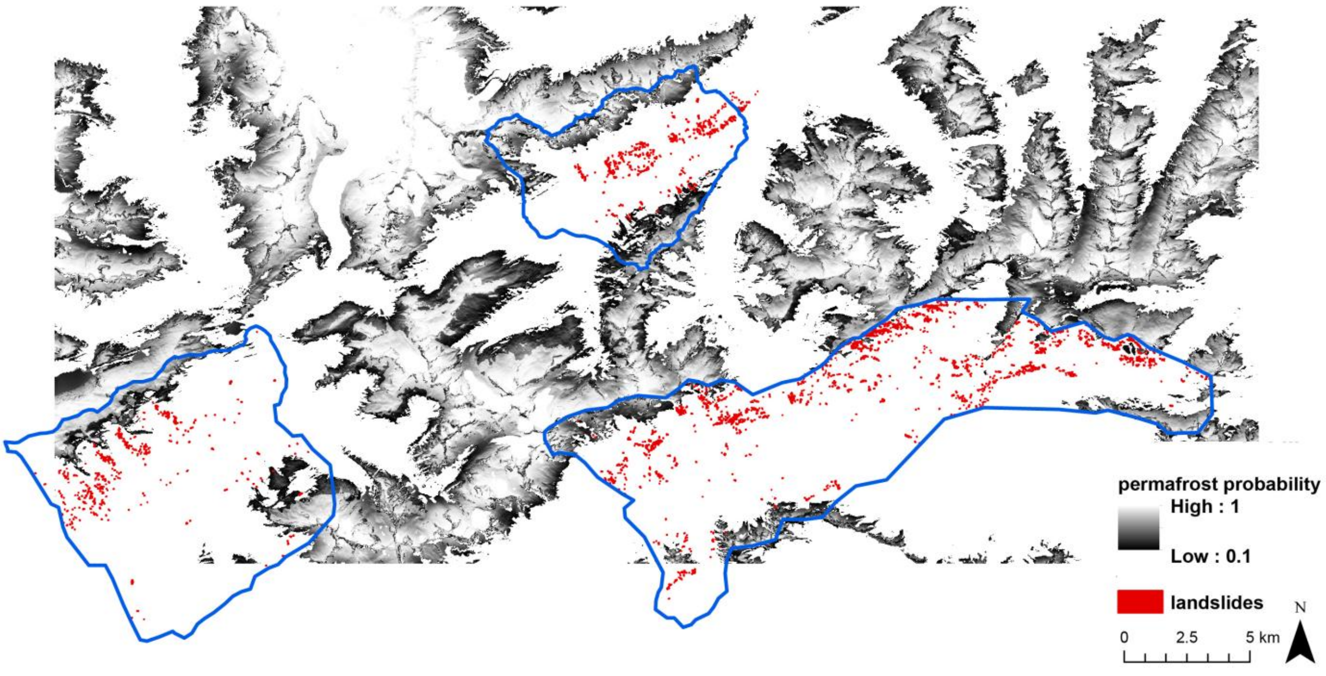
Land sliding is also a process with causal links to climate change, primarily through precipitation, but in some cases also indirectly (as discussed in chapter 2.1.2) through temperature (Crosta and Clague 2009). These changes in temperature and precipitation trigger secondary effects associated with landslide susceptibility: changes in the extent of glaciers and the distribution and duration of the snow cover and permafrost (Stoffel and Huggel 2012). From the theoretical understanding, an increased mass-movement activity as a result of predicted climate change in mountain environments is likely. However, observational records of changes in activity are scarce (Stoffel and Huggel 2012) and scenario-driven global predictions are highly uncertain due to the lack of spatial resolution (Crozier 2010). The observed impact of climate change on landslides is typically ambiguous and physical cause-effect relationships often remain speculative or conceptual (Huggel et al. 2012). However, the monitoring of recent large landslides in some mountain regions suggests that climate change is having an effect on slope stability (Evans and Clague 1994; Geertsema et al. 2006; Jakob and Lambert 2009).

In the elevated areas of the Bedretto/Piora valleys, where anthropogenic effects are absent or minimal and where snow, glaciers and permafrost, which are sensitive to temperature changes, occur, slope stability may be affected mainly by climatic effects. Climate change may affect rates of physical and chemical weathering or degrade permafrost, which may change the bulk strength properties of slope-forming materials (Arenson and Springman 2005; Harris et al. 2009).

Degradation of permafrost is a potential candidate responsible for the observed increase in shallow landslides in the Bedretto/Piora valleys and maybe also in the Obergoms and Urseren valleys. Permafrost exists in many steep, north-facing slopes in high mountain environments. Permafrost degradation and slope stability are related in a complex way, and the corresponding research field is relatively young (Gruber and Haeberli 2007).

However, an overlay of the landslide inventory maps with the permafrost probability map (Fig. 37) indicates hardly any overlap. Indeed only less than 1% of the landslides are located in potential permafrost areas and even though the number has increased over time, it cannot account for the strong trends. More likely, changes in snow cover and dynamics are responsible for the recent increase in shallow landslide occurrence in the Bedretto/Piora valleys. While many studies are concerned with rainfall as a trigger for shallow landslides, snow as a triggering factor is scarcely investigated and further research should be dedicated to this topic.

Fig. 37 > Location of shallow landslides (red) with respect to the permafrost probability map (Boeckli et al. 2012)



Taken together, climate change may increase landslide activity in mountain areas. This has also been shown in our studies for the Obergoms and Bedretto/Piora valleys (see above). However, more observational data are needed to document the recent development trends. With respect to the long-term development, geomorphologic feedback mechanisms need to be considered. Susceptibility may again decrease over time due to the “emptying” of the slope (Hufschmidt et al. 2005) or may further increase over time due to a steepening and undercutting of slopes (Claessens et al. 2007). Which process dominates will most likely vary from region to region.

Last but not least, if farmers react to climate change by implementing different crops or, with greater relevance for alpine areas, change land-use patterns, the soil redistribution patterns and even landslide susceptibility (as discussed below) may change. This effect might be very strong if we consider that predictions suggest the possibility of rainfall changes of the order of a few percent in rainfall amounts and intensities while changes in land cover and land use can be much more efficient. For instance a reduction in vegetation cover from 100 to 0% resulted in an erosion yield more than 250 times higher (Schindler Wildhaber et al. 2012).

4.2 Effects of land-use change

A main agricultural development that influences erosion susceptibility is the abandonment of remote pastures. Since the late nineteenth century, the abandonment of agricultural sites, especially the remote high alpine areas, is a well-known phenomenon (BFS 2001; Descroix and Mathys 2003; Piégay et al. 2004; Lasanta et al. 2006; Tasser et al. 2007; Alewell et al. 2014). Simultaneously, the remaining farmland in lower areas with high accessibility is used more intensively (Meusburger and Alewell 2008). In the Swiss Alps, for example, the area of summer pastures has steadily decreased (from 612 619 ha in 1954 to 465 519 ha in 2005) (Troxler et al. 2004) due to abandonment and subsequent emergence of shrubs and reforestation. Simultaneously, the stocking numbers have increased, resulting in intensification in the form of increased stocking rates, extended grazing periods, more heavy pasture animals and higher fertilizer inputs mainly on accessible areas. In the Swiss High Alps, the livestock population has increased from 200 000 to 420 000 sheep during the last 40 years (Troxler et al. 2004). Furthermore, the land-use management has changed: permanent herding of cattle and sheep has been mostly abandoned since 1950 and has been replaced by uncontrolled grazing. The latter has resulted in a significantly higher grazing intensity of high alpine meadows (Troxler et al. 2004). Thus, even though animal numbers and grazing intensity have increased considerably in some areas, many other areas are affected by shrub encroachment, because the animals are not herded and maintenance of the sites has decreased (Caviezel et al. 2010).

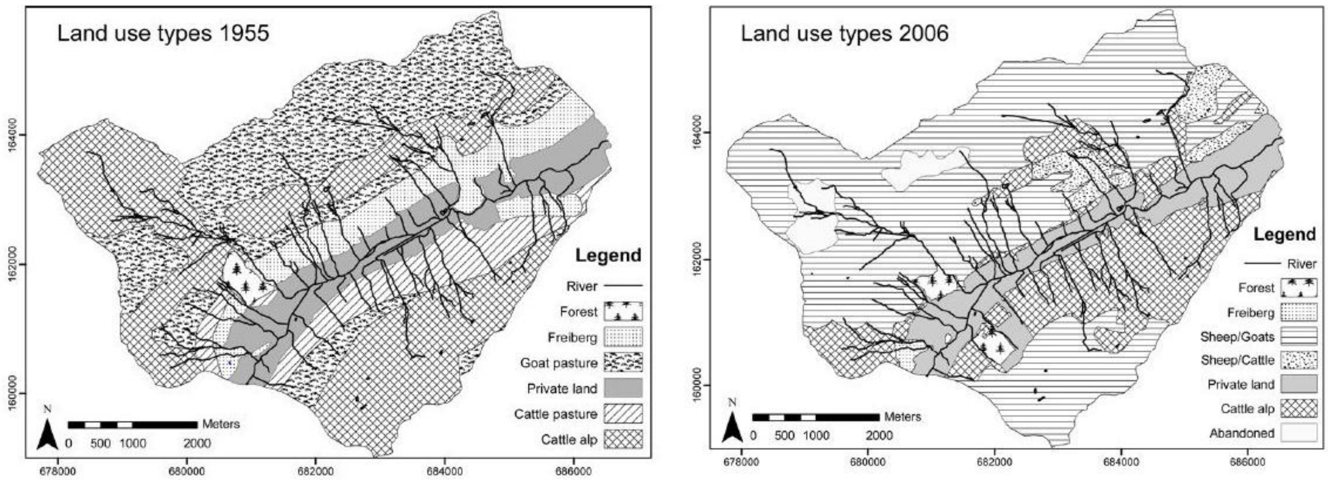
The Urseren Valley is a good example of the above described land-use development. During the last decades, land-use has intensified in the valley, which is shown by the decreasing pasture area per animal (Fig. 38).

Land-use management practice is a further factor that has changed over time. The change of management practice is apparent from the comparison of pasture maps of 1955 and 2006 (Fig. 38). The private land of the farmers is situated at the valley floor and adjacent slopes. These areas are mainly used as hayfields. Different forms of pastures (goat, cattle and sheep) are located on the slopes. The following developments are noticeable from the pasture maps of 1955 and 2006:

a) Goat pastures disappeared in 2006.
b) The traditional land-use type called *Freiberg* almost disappeared. *Freiberg* areas are pastures which are used in spring because of the vicinity to the farms and the more advanced vegetation state at this altitudinal level. The appointed date on which cattle are brought to the higher pastures is 14 June. For the rest of the summer the *Freiberg* is kept as a reservoir in case of an early onset of winter and left to regenerate during the main growing season (note that the *Freiberg* areas were situated in the geologically sensitive area of the Mesozoic layer).
c) Remote and less productive areas were abandoned.

d) Alpine cattle alps, which are high mountain pastures used exclusively during summer, disappeared completely and are now sheep pastures.

Fig. 38 > Pasture maps for the years 1955 and 2006 for the Urseren Valley



Meusburger and Alewell 2008

Farmers were interviewed concerning which kinds of land-use changes have happened and where changes in land-use intensity have occurred since 1955. The observations and experiences of the farmers can be summarized in two general developments: (i) an intensification of the areas close to the valley (points 1 to 6, see below) and (ii) an extensification of remote areas (points 7 to 9, see below). The developments are ascribed to the following agricultural changes, which agree with developments described for other alpine regions (Tasser and Tappeiner 2002; Troxler et al. 2004; Mottet et al. 2006; Tappeiner et al. 2006; Baur et al. 2007):

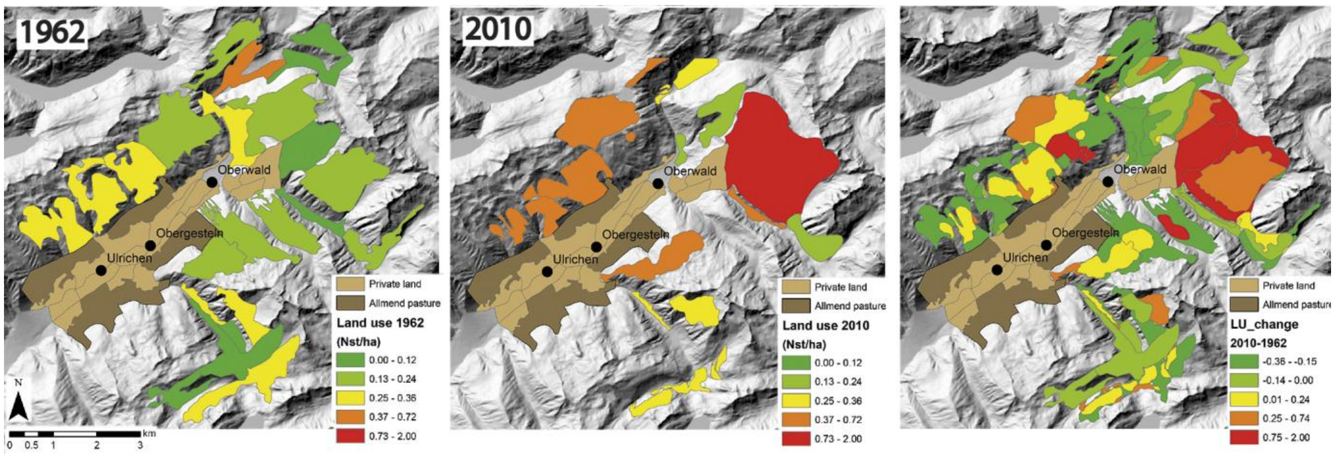
1. Permanent herding was replaced by uncontrolled grazing within fenced pastures, where animals are concentrated at single locations, for example at the watering place. It is typical for permanent pastures that the pasture intensities are heterogeneously distributed due to the influence of topography and the location of the water sites.
2. Steep areas that are difficult to access and that have formerly been mown by hand (wild haying) were converted to pastures.
3. The number of external pasture animals delivered from the lowlands outside the valley increased. The animals are delivered on an appointed date independently of the actual vegetation status in the area.
4. Compulsory labour that was used for the maintenance of the pastures was abolished.
5. The *Freiberg* land-use type was largely abolished. The areas are now pastured throughout the entire summer season.
6. The private land is nowadays more frequently mown with machines and intensively fertilized with organic manure. Especially the addition of organic fertilizers may be a reason for the enhanced landslide susceptibility, because it favours species with flat rooting (von Wyl 1988).

7. Stocking with dairy cows on the alps was reduced because of increased mother cow husbandry. The latter is mainly carried out at the valley floor due to difficulties in herding and a more frequent need for veterinarian assistance.
8. The shrub cover increased due to the cessation of wild haying and firewood collection and reduced stocking of the alps with cattle and goats. Goat grazing especially hampered the invasion of shrubs (Luginbuhl et al. 2000).
9. The number of farmers declined (e.g., in the Urseren Valley, from 77 in 1970 to 31 in 2006). Simultaneously, there are fewer full-time farmers.

Land-use development in the Obergoms Valley

In the Obergoms Valley we observe similar agricultural developments. However, the spatial pattern of land-use intensification and extensification is more complex.

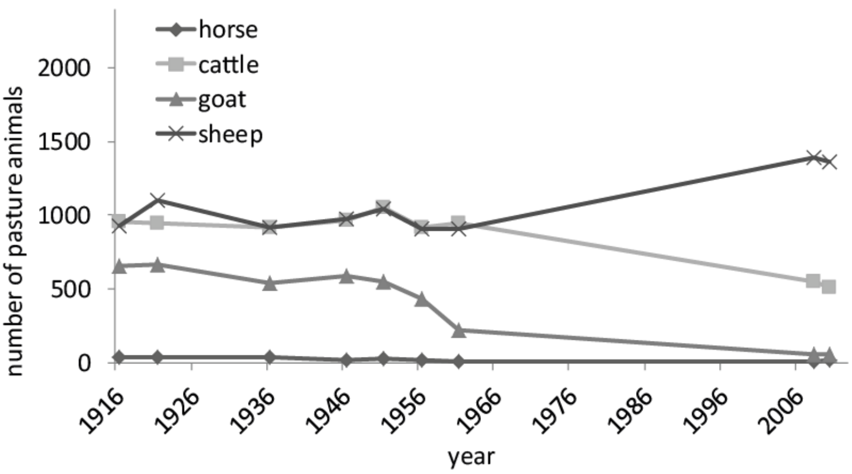
Fig. 39 > Land-use maps of the Obergoms Valley for the years 1962 and 2010 and the land-use change map resulting from the difference between the stocking intensities of the two years



Traditionally the land-use is divided into three sections: the private agricultural land at the valley bottom (Fig. 39, light brown), the *Allmend* pastures on the lower slopes (Fig. 39, dark brown), and the alp pastures on the higher slopes (Fig. 39, coloured). *Allmend* (“commons” in English) refers to land accessible to all members of the community.

The private agricultural land at the valley bottom is mainly flat and as such easy to cultivate with machines. In addition, these areas have a high economic output because they belong to an agricultural zone which is strongly subsidized. The upslope areas formerly used as hayfields are now used as *Freiberg*, which means that they are pastures before the cattle are brought up to the alpine pastures and after the summer grazing. The *Allmend* pastures adjacent to the private agricultural land consist mainly of forest pastures. On the north-facing slope, where the forest stocking was always denser, the use of the areas was very extensive and was abandoned over time. At the altitude of 1500–1800 m a.s.l. these *Allmend* pastures are hardly used these days, and due to the fact that the land is not owned privately there will be no subsidies for these areas (F. Hallenbarter, personal communication). Moreover, this altitude was always forested for avalanche protection. The alpine pastures are located upslope on forest-free grassland. The farmers have been subsidized according to the number of animals that are pastured over the summer. In order to quantify the effect of grazing of different animals, the livestock unit is used, which gives the grazing equivalent of a pasture animal compared to one adult cow. This number is weighted with the number of pasture days to derive a *Normalstoss*. *Normalstoss* is defined as the pasturing of one large livestock unit for 100 days. Thus, to derive the number of *Normalstoss* the large livestock unit is multiplied by the number of pasturing days. The grazing equivalent of different animals was converted to large livestock units differently in 1962 and 2010. Thus, for the purpose of comparison of the pasture maps, both were produced based on the conversion factors of 2010.

Fig. 40 > Development of the total number of different pasture animals for the community of Oberwald, Obergesteln, and Ulrichen in the Obergoms Valley



data according to Imboden 1962

The total number of livestock units increased by only 3.2% since 1962. However, the composition of pasture animals changed (Fig. 40). The number of sheep increased from approximately 900 to 1300 while the number of cattle declined from approximately 9520 to 550. Especially, the importance of goat pasturing already declined after the year 1946. Previously, the number of goats varied around 500 to 600, but now there are only 50 to 60 goats left. A problematic issue of the alpine pasture use was that the pastures were exchanged between farmers every year. As a result of this rotation, the farmer felt less responsible for the maintenance of the land and infrastructure. Nowadays the farmers own the alpine pastures.

The data of the two valleys confirm that changes in alpine grassland management differ considerably locally and affect erosion risk and landslide susceptibility.

Consequences for landslide susceptibility prediction

The dynamic nature of soil erosion risk factors causes problems for the prediction of soil erosion risk due to shallow landslides. Unconstrained by the statistical method chosen, the basic concept of landslide hazard assessment with statistical methods is to compare the conditions that have led to landslides in the past with the conditions in regions currently free of landslides (Carrara et al. 1998). In other words, the assumption made when using multivariate statistics for landslide hazard prediction is that catchment characteristics leading to landslides in the past will also be susceptible to landslides in the future.

The relation between past and future may, however, be considerably weakened when landslide causal factors become variable with time (for example due to land-use and climate change). The result may be unsuitability of landslide susceptibility maps to predict future events under changed conditions (Guzzetti et al. 2006).

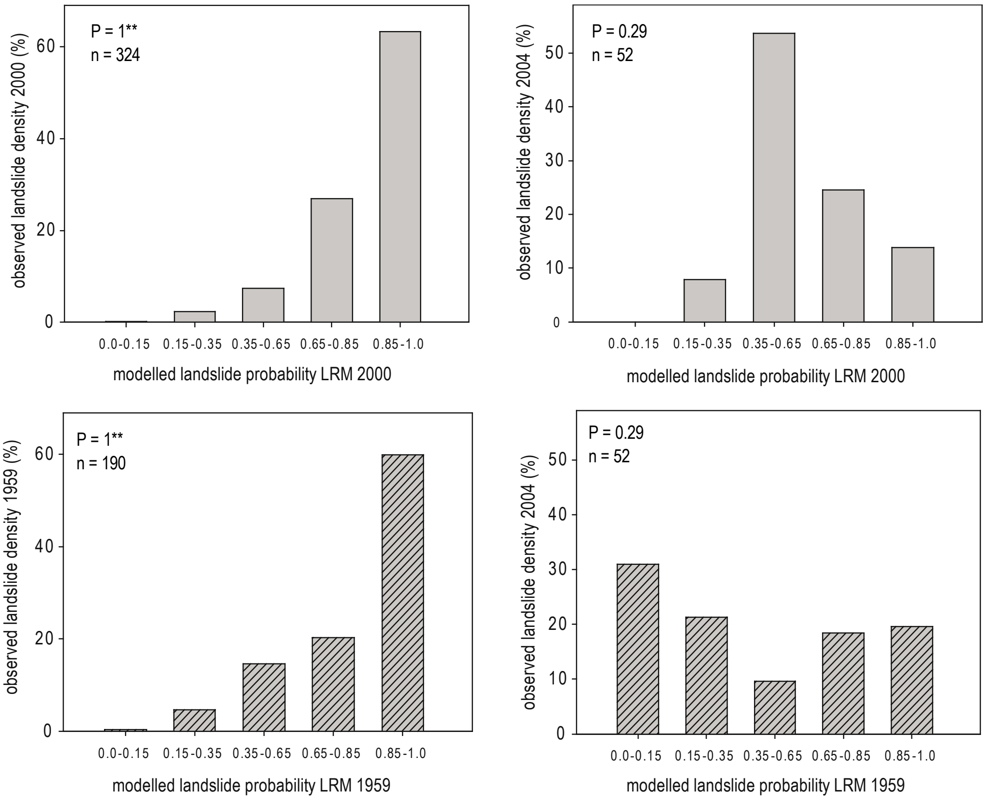
The latter was confirmed for the landslide susceptibility map of the Urseren Valley, because even though the performance of the susceptibility map according to classical criteria was good (see ROC curve and modelled susceptibility versus predicted landslide susceptibility), the model could not predict the new landslides that occurred in 2004 outside the calibration window used (Fig. 41, right).

Compared to the quantity of landslide susceptibility studies, relatively little effort has been made to validate the predictive capability of the obtained maps (Chung and Fabbri 2003) and even less to generate maps of likely future landslide scenarios (Zêzere et al. 2004; Guzzetti et al. 2006; Irigaray et al. 2007). The latter was done for instance by integrating the susceptibility map with the return period of rainfall (Zêzere et al. 2004). However, for this approach the spatial distribution of susceptibility zones remains unchanged and a change of the spatial susceptibility pattern over time is not considered (Zêzere et al. 2004). A strong element of uncertainty is introduced when the importance of landslide causal factors changes rapidly. For instance, human action – mainly land-cover and land-use changes – may increase the sensitivity of the geomorphic system to triggering events and thus cause a shift of susceptibility zones. This negligence may impede the identification of new potential susceptibility areas and hence may hamper the timely initiation of prevention measures.

Temporal validation adds a time element to susceptibility maps and makes the transfer to a landslide hazard map possible (Remondo et al. 2003). However, when land-use and climate changes occur, the susceptibility map fails to predict the new landslides. Thus, another approach was tested in order to introduce a temporal component to the susceptibility map.

Fig. 41 > Receiver operating characteristic (ROC) curves for the two logistic regression models (left) and dependency of modelled landslide susceptibility and actual landslide densities (Spearman's rank correlation coefficient, P; correlation is significant at the 0.01 level)

The landslide density values are based on 324 mapped landslides for the year 2000 (upper part), 190 mapped landslides for the year 1959 (lower part), and 52 mapped landslides for the period 2000–2004 (right)

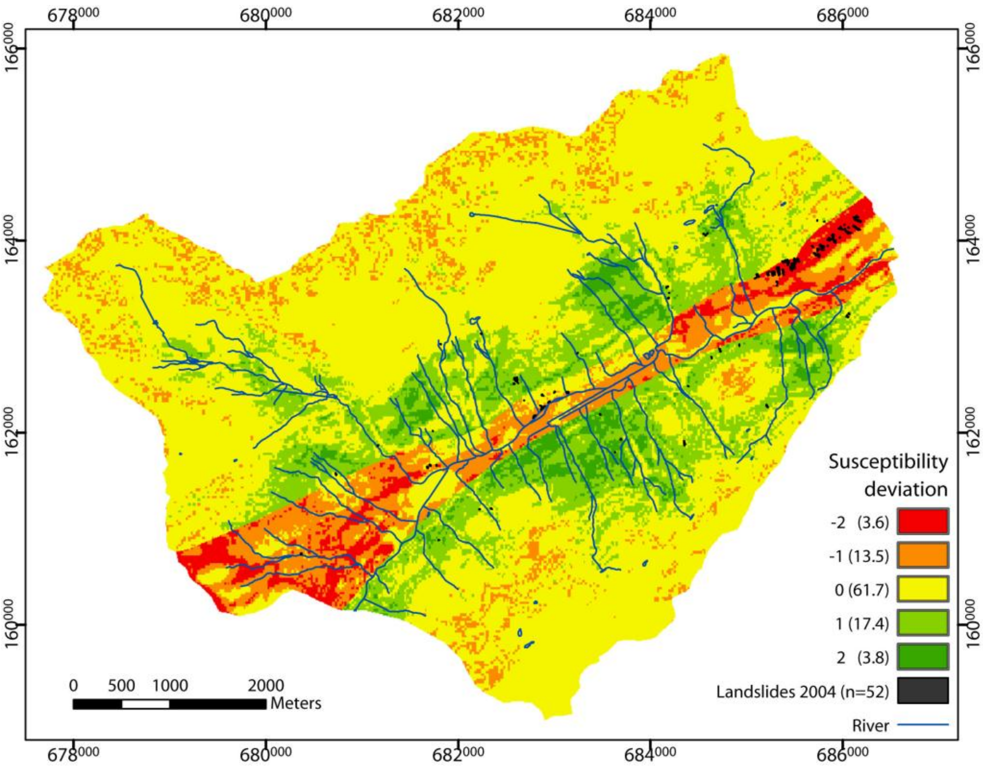


Meusburger and Alewell 2009

The assumption made is that if recent landslides (2004) are a result of changing land-use and climate patterns and not an accidental deviation of the predicted probability, the change of susceptibility over time should be related to the new landslides. The landslide susceptibility change map was generated by subtracting the modelled susceptibility map of 2000 from the susceptibility map based on data from 1959.

The resulting values are negative where the landslide susceptibility was lower (indicating lower landslide probability) in 1959 and positive where it was higher. The landslide susceptibility mainly increased near the valley bottom and the adjacent lower slopes. In contrast, a decrease is visible for the more remote slopes except for the higher elevated areas with high rock and debris contents, which show unchanged susceptibility.

Fig. 42 > Differences in susceptibility zones between the logistic regression models of 1959 and 2000 (Projection: CH1903 LV03)



Meusburger 2009

This deviation susceptibility map shows good correspondence with the new landslides that predominantly occurred at the foot of the slope after 2000 (Fig. 42). About 85% of the new landslides occurred in the zone with the highest susceptibility deviation (value: –2) towards an increased landslide probability over time. With this multi-temporal data analysis the temporal shift of susceptibility zones could be spatially captured and visualized. Thus, it is possible to add a spatially explicit time element to susceptibility maps in order to improve the assessment of future landslide susceptibility zones.

Further, the pattern of the deviation zones may give information about the landslide causal factors that caused the shift of susceptibility zones and maybe also the increasing trends in landslide occurrence.

The most plausible explanation for a local shift in susceptibility zones is the change in land-use types between 1955 and 2006 (Fig. 38). New high susceptibility zones (with a deviation of -2) are located within two land-use types (*Freiberg* and private land). For these two land-use types, land-use intensified during the last decades. For all other land-use types, no distinct shift of landslide susceptibility zones is evident.

5 > Mitigation of soil erosion by careful land-use management (by Jakob Troxler)

Even though extreme rainfall will always trigger soil erosion, careful pasture management can reduce the amount of soil transported from the mountains to the valley.

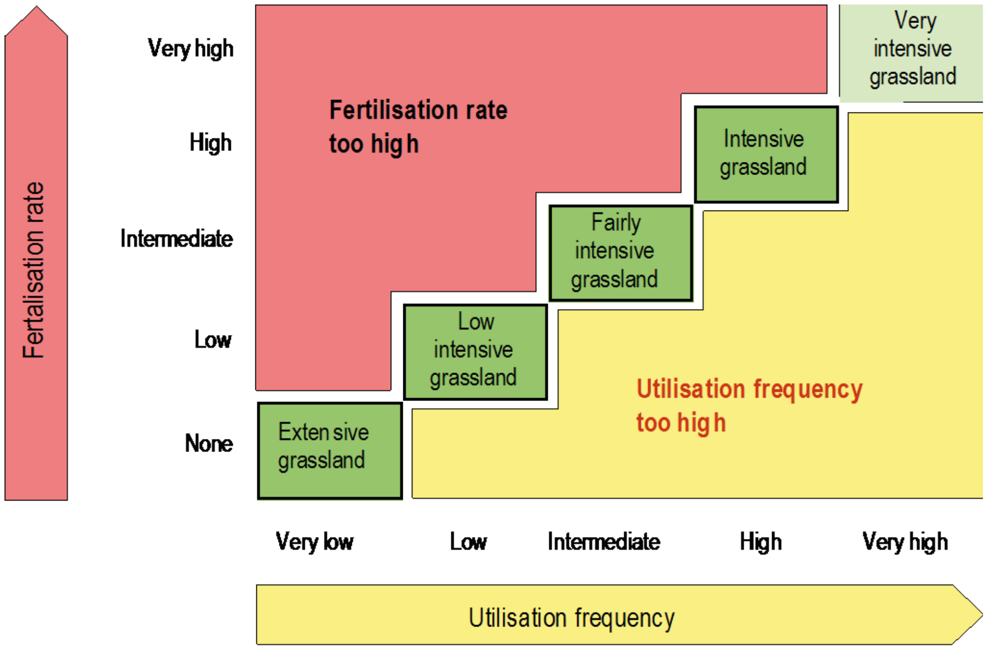
Some of the most important parameters having an influence on whether or not the sward is intact are animal species, animal behaviour, animal live weight, stocking rate and density, pasture system and fertilization. A dense vegetation cover with high biodiversity is supported by appropriate pasture management. As pointed out in previous sections, an intact and diverse vegetation cover can significantly reduce erosion risk and soil loss. Traditionally, alpine dairy farming promotes a balanced, productive vegetation cover of high quality and is interested in the long-term preservation of the managed grassland.

Soils in mountain areas are often shallow and characterized by a low supply of nutrients. Thus, fertile soil is a valuable resource and a productive basis for mountain agriculture. Soil erosion negatively affects several soil functions; for instance the reduction of water storage capacity may restrain plant growth or even change plant communities. As a result, the productivity of a site declines. Furthermore, large-scale soil erosion (both sheet erosion and landslides) changes the landscape and scenery, and the restoration of damages is costly, particularly in difficult-to-access alpine regions (Sutter and Keller 2009).

Utilization and fertilization

With respect to the environmental conditions, a good balance between utilization and fertilization intensity should result in a stable, balanced, and site-appropriate vegetation cover with a dense sward (Fig. 43). This leads further to a good soil structure and soil activity.

Fig. 43 > The main types of grasslands according to the management intensity



Sahli et al. 1996

The manure produced on an alp needs to be disposed of close to the stables. Deeply developed soils allow higher fertilizer input than shallow soils. Thus, shallow soils should receive the faeces and urine of the pasturing animals only. Difficult-to-access sites will generally not be manured. If the fertilization input is too high compared to the utilization frequency, the sward will loosen and the susceptibility to the mechanical impact of trampling and machines will increase.

Mechanization

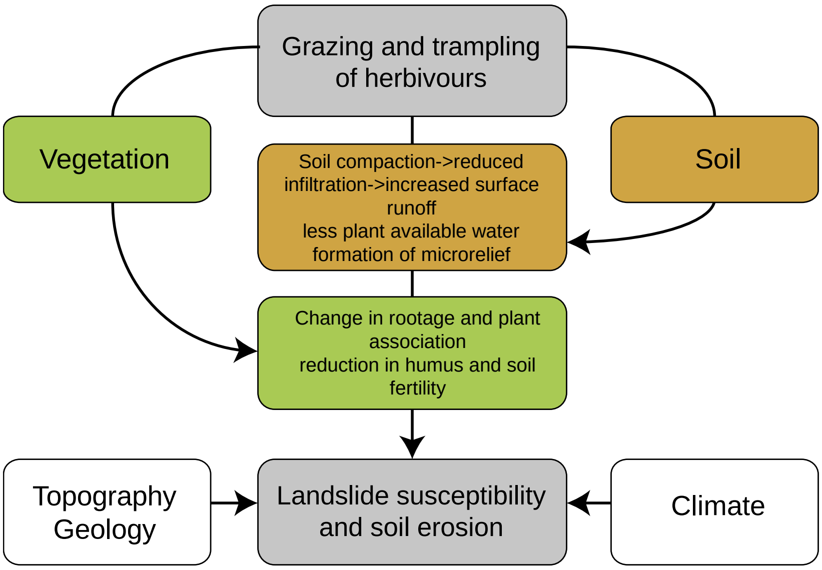
During the last years, the use of transport and harvest machinery specifically for mountain grasslands increased. The type of tyre and low tyre pressure may reduce the damage to the vegetation, soil compaction, and thus soil erosion. Depending on the farming needs, it is important to choose the correct machinery for each site.

Farms that operate year round produce enough farmyard manure to meet their fertilization demand (farm gate balance) (Jeangros and Troxler 2006). Modest fertilization of rather low-nutrient meadows with phosphorous and potassium increases the proportion of leguminous crops, which improves the feed quality and availability of feed over the season and increases the elasticity of utilization. However, this requires the farmer to have a good knowledge of vegetation dynamics as well as animal husbandry.

Influence of herbivores on soil erosion

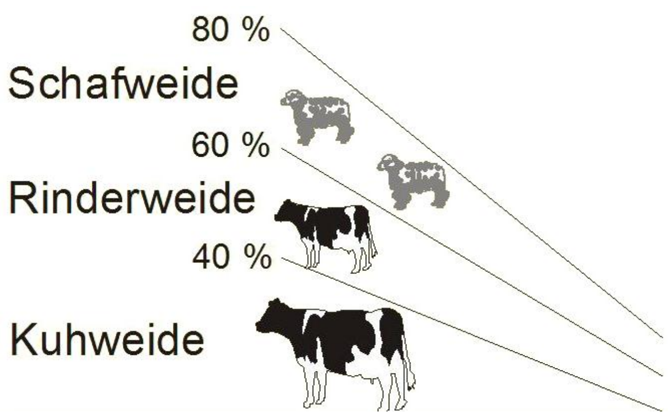
Pasturing animals influence soil erosion mainly via foot pressure or trampling and by the reduction of plant material and selective grazing of plant species (Fig. 44).

Fig. 44 > Influence of herbivores on soil erosion



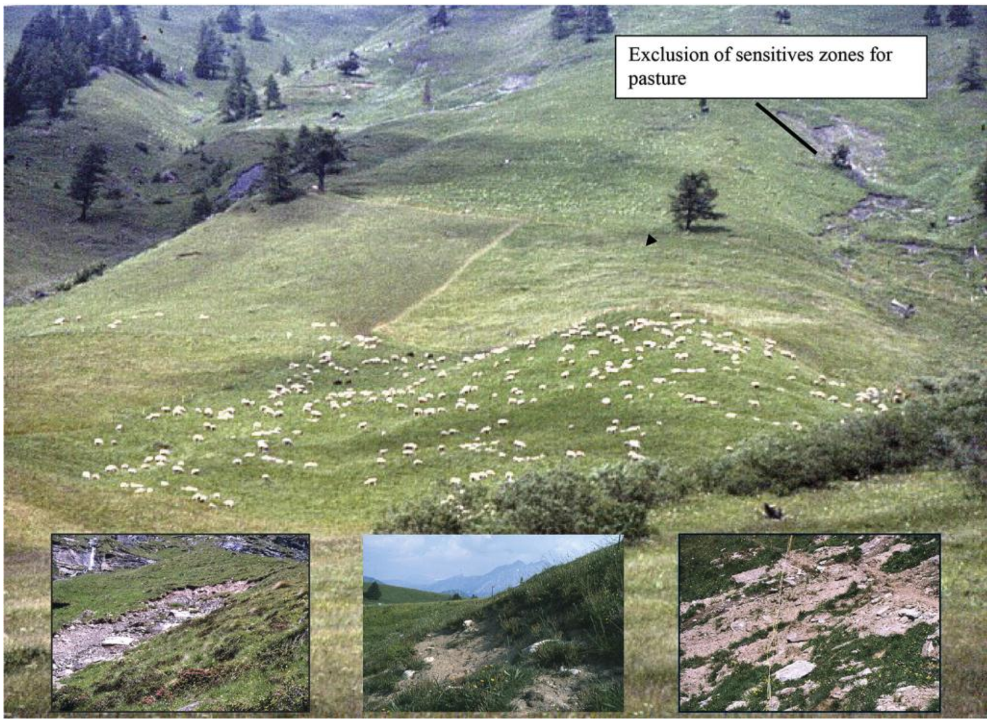
The choice of the pasture animal should depend on the slope angle of the pasture (Fig. 45). Heavy cows should only be pastured on areas with a maximum slope angle of 40% (18°). Only young cattle should be kept on areas with slopes between 40 and 60% (27°). On even steeper slopes, only small livestock animals like sheep and goats are recommended. Very steep slopes need to be excluded from pasturing. In fact the pressure of a cow's claw is about three to five time greater than that of a sheep's claw and causes greater footstep damage. For this reason, lightweight species should be favoured in mountain areas.

Fig. 45 > Suitability of pasture animals depending on the slope angle of the pasture



Especially during wet weather conditions, there is a need to provide enough pasture and fresh grass to avoid unnecessary trampling by the animals in search of food. Trampling on wet surfaces causes immense damage to the vegetation and soil within a short time. A further option is to reduce the stocking rate. Areas already affected by soil erosion should be excluded from the remaining pasture area by fences. In order to receive subsidies for summer grazing, fencing and animal rotation are required (Fig. 46).

Fig. 46 > Correct rotation grazing with sheep and (below) erosion features caused by pasturing



Animal behaviour

The different behaviours of pasture animals may affect the soil and the vegetation and thus need to be considered. For instance sheep walk more than cattle, and thus the development of trails is more likely. Trails are potential tear-off lines for landslides. Therefore, there are good reasons to favour sheep: sheep claws put less pressure on the ground than cows and trails can be avoided with effective pasture management. However, it needs to be considered that sheep prefer higher locations of a pasture, while cattle prefer the lower parts because they are less active. Thus, the composition of herds with different species needs to be adapted to the pasture characteristics. Pastures with high inclination are unfavourable for horses, which like to gallop. However, compared to cattle, sheep, and goat, horses can cope very well with strong mat-grass pastures (with, for example, a high percentage of *Nardus stricta* in the sward, which is a very sparse pasture with poor fodder).

Stocking rate

Reduction of the stocking rate is a good strategy to avoid soil erosion, but the reduction enhances the risk of shrub encroachment. Even though the total number of sheep has more than doubled during the last 40 years, on several alpine pastures the stocking rate has been reduced, which leads to unintended shrub encroachment and reforestation. The latter is due to uncontrolled grazing with neither herding nor fencing and generally less management intensity. Rotation grazing is a good way to minimize the risk of shrub encroachment, ensuring homogenous pasture intensity. Several studies showed that with an adapted management of pastures, considering altitude, vegetation unit and slope aspect, vegetation quality can be kept at a high level and it might even be possible to achieve beneficial effects regarding the individual performance of the animal (Chassot and Troxler 2002).

Pasture systems

Generally three mountain pasture systems are practised: continuous grazing, strip grazing and rotation grazing, of which the last is the most efficient system.

Continuous grazing or free grazing is a one-pasture system in which livestock have unrestricted access to the pasture area throughout the entire grazing season. The extensive continuous grazing system has the advantages of little labour effort and low material expenses for the construction of fences. Moreover, the allocation of large pasture areas allows for an early start of the pasturing season in spring, with little damage through trampling during wet conditions. However, the disadvantages of the system can be severe: irregular pasture intensity, development of over- and undergrazed areas, and development of cattle trails in heavily used areas. In low-pasture-intensity areas the fodder quality declines and subsequently shrub encroachment and later reforestation occur.

Fig. 47 > Pasture management with shepherding: permanent control of sheep movement (left), free grazing system with sheep in high mountain pastures (middle and right)

The root mass and root depth of the plant will decrease in overgrazed areas. As a consequence, the vegetation will be more susceptible to dehydration and trampling damages. Particularly, sheep should never be kept using a free-grazing system, because due to their preference for elevated areas with young grass, the higher parts will be eroded and the lower parts will inevitably deteriorate (Fig. 47, right).

Nowadays, light and cost-effective fence material for small livestock is available. For large herds of sheep (or goats), shepherding is becoming more important. Shepherding can replace fences, but the effectiveness of the management is largely dependent on the skills of the shepherd (Fig. 47, left). Herd control by shepherds only has a positive influence on the sward if pasture sectors are regularly changed.

Rotation grazing, also known as cell grazing, is the optimal pasture system for mountain areas. It is a paddock system in which herds are regularly and systematically moved to fresh, rested areas with the aim of maximizing the quality and quantity of fodder growth. It allows for short stocking and long regeneration periods (BAFU and BLW 2013). The result is that there is little damage due to trampling since the animals stay on the pasture for a restricted time period. Irreversible damage to the sward and the soil structure can thus be avoided. A high sward density is also promoted because the excrement of the pasturing animals is distributed more homogeneously. The number of paddocks depends on the desired pasture intensity, the homogeneity of the vegetation and the topography (slope aspect and relief). The differences in elevation within a paddock should be low for sheep, otherwise the higher parts will be overgrazed (Fig. 47, right).

Strip grazing is a grazing management system that involves providing livestock with a fresh allocation of pasture each day. It is usually organized within a paddock grazing system and the animals are controlled by the use of an electric fence. While this system has the advantage of maximal fodder use in the optimal state (which is important for lactating animals), it also demands continuous and daily observation, maintenance and control of the fodder availability. Extreme damages may result if the fodder supply is too low.

Mixed species grazing

Mixed-species grazing means two or more animal species graze together at the same time on the same grazing area. The rationale for mixed-species grazing is based on the principle that animals have different grazing preferences. Thus, mixed grazing promotes a homogeneous pasture intensity, which leads again to better fodder quality and higher live-weight gain per animal and live-weight gain per unit of area. This is also partially due to additional benefit from better parasite control. Sheep, cattle and horses are generally affected by different gastrointestinal parasites, whereas sheep, goats, and camelids share the same parasites. Another advantage of mixed grazing can be that goats graze on shrubs. Thus, if goats are pastured together with cattle or sheep, shrub encroachment is reduced. Horses are also valuable for mixed grazing because they have different fodder preferences, such as for *Nardus stricta*. However, the pastures should not be too steep.

Stable or no stable

Since the Second World War, central stables for lactating animals have been favoured as they require a lower capital investment. On large alpine pastures with considerable differences in altitude, the distances between the pasture and the stable are long and cause trails. Trails enhance both water-induced erosion and landslides (Fig. 48). The utilization of mobile milking parlours can largely solve this problem (Troxler et al. 1992), because the milking parlour is brought to the animals. Well-constructed platforms are needed at the milking places to minimize trampling damage. The mobile milking parlour has been adopted in many areas with positive results. In most cases a stable is not required for young cattle. Studies on dairy cattle farming indicate that dairy cattle can handle the summer pasturing without a stable (Jans and Troxler 1992).

Fig. 48 > Long distances between stable and pastures: formation of trails

To sum up, soil erosion can be minimized in various ways. Even though the utilization of machines that are well adapted to mountain slopes is important, even more attention should be paid to the choice of animal species and animal type. Heavy cows with high performance are usually not adapted to good management of alpine pastures. The balance between utilization and fertilization level is very important. Moderate fertilization results in a better root density in mountain regions. Year-round mountain farming allows a good farm gate balance through the optimal use of organic fertilizer produced on the farm.

Pasture with rotation grazing systems and good pasture management is the key to a compact and dynamic sward. Not maximal but moderate stocking rates give the best animal performance. Mixed grazing pasture systems (sheep, cow, goat, horse) are recommendable and should become more important in the future.

6 > Reporting on two workshops on "Soil Erosion in the Alps" (Andermatt 2007, Basel 2012)

6.1 Workshop in Andermatt, 2007

Aim of the workshop

The University of Basel, the Environmental Protection Agency of the Canton Uri, and the Federal Office for the Environment (FOEN) organized a scientific workshop and information event related to soil erosion and degradation in the Alps. During the workshop, which was held in Andermatt, scientists and stakeholders discussed scientific results and perspectives on soil erosion in the Alps. Beside the scientific view of causes and future development of alpine soil stability, the views and perspectives of the canton, the Federation, and the EU were addressed. Moreover, experiences from other cantons were exchanged.

The workshop started with an excursion to the test sites of the University of Basel. The workshop participants were given an overview of the soil erosion processes relevant to the Urseren Valley. Afterwards the scientific workshop was opened by B. Bühlmann from the Environmental Protection Agency of the Canton Uri. The Canton Uri was introduced to the audience, with special focus on the threats for the resource soil in the canton. A major threat to soils originates from extreme rainfall events and subsequent flooding, landsliding and soil erosion. Another important focus is on the constructional activities in connection with the New Railway Link through the Alps (German: Neue Eisenbahn-Alpentransversale, NEAT) and touristic infrastructure like ski pistes and, for instance, the Sawiris project in Andermatt.

Scientific workshop

J. Zihler from FOEN highlighted the importance of applied research in order to create a solid scientific foundation with regard to legislation. While in the past the focus was mainly on chemical soil protection, in the future, physical soil protection especially in alpine areas will gain importance.

After these introductory words the scientific discussion started with J. Troxler from the Agroscope Changins-Wädenswil, who explained the different grassland management options for alpine areas. Moreover, he commented that the soil erosion features observed during the excursion in the Urseren Valley are partially due to mismanagement.

R. Wyss on the other hand discussed the high natural susceptibility of the Urseren Valley to soil erosion and landsliding, which is due to its geological situation.

G. Markart from the Federal Research and Training Centre for Forests, Natural Hazards, and Landscape, Innsbruck, Austria, stressed again the impact of alpine grassland management for runoff generation and subsequently soil erosion. His very convincing data were based on several rainfall experiments conducted in different areas of the Austrian Alps and the South Tyrol, Italy.

E. Hiltbrunner from the Botanical Group of the University of Basel investigated the effect of vegetation on soil erosion in the upper part of the Urseren Valley close to the Furka pass. Her results clearly indicate the importance of the fractional vegetation cover for soil erosion. Different biodiversity indicators were not correlated to soil erosion. However, different functional groups can affect surface runoff. Moreover, her data pointed to the slow regeneration rate of alpine vegetation. Taken together, all these talks highlighted the importance of vegetation cover for sheet erosion.

Regarding the interaction of vegetation / land use and landsliding, E. Tasser from the European Academy (Bolzano, Italy) presented data from five alpine sites located in Austria and Italy. Besides topographic factors, life-form types are another group of factors relevant for landslide susceptibility. According to his results, increased coverage with grass and herbs leads to a decrease in landslide susceptibility. But as the coverage with dwarf shrubs increased, the landslide susceptibility also increased. An increase in total root density seems to reduce the probability of landslides. Moreover, the risk of soil fractures decreased even further due to higher root density exactly at the average fracture depth of the landslides. The highest root densities can be found in extensively and traditionally used mixed grassland communities (*Hypochoero-Nardetum, Caricetum sempervirentis*, *Larix decidua* (initial stage)), and the smallest in abandoned areas, especially in dwarf shrub communities (*Junipero-Arctostaphyletum*) and forests (*Larici-Piceetum*), but also in very intensively used hay meadows in the valley bottoms. In conclusion, the highest landslide susceptibility is expected in abandoned and in intensively used sites if only root density is considered.

P. Bebi from the Institute for Snow and Avalanche Research (Davos, Switzerland) tried to answer the question of whether reforestation is a virtue or a curse with respect to CO_2 emissions, biodiversity, and tourism and soil erosion. For the latter, he concluded that several questions about the impact of forest on soil erosion still need to be resolved. Particularly, the potentially instable conditions during succession from grassland to forest need more attention. Overall, a forest certainly has a stabilizing effect but this effect is strongly dependent on the specific site conditions.

With M. Lehning, also from the Institute for Snow and Avalanche Research (Davos, Switzerland), the topic was changed towards snow and ice and the question of which research topics remain in the context of soil erosion. Also, L. Braun from the Commission for Glaciology (Munich, Germany) reported on the effects of climate change on glacier runoff and bed-load discharge.

Informative meeting

The second day of the workshop was dedicated to natural hazards and started with introductory words from M. Stadler, the *Landammann* and Vice director of the Canton Uri and A. Imhof from the Environmental Protection Agency of the Canton Uri. The Canton Uri is always confronted with natural hazards and erosion, and increases are expected in the future mainly due to climate changes. Uri aims at prevention of soil erosion and natural hazards rather than subsequent remedial maintenance. An important question for the canton is: How much of the erosion is caused by anthropogenic interference and how much is natural? In this context it is very important to investigate the question of which land-use management is sustainable for the soil and slope stability. C. Böbner, Vice Director of the Federal Office for Agriculture, is interested in the preservation of mountain agriculture and as such 61% of all direct payments (total: 2.5 bn CHF) is dedicated to the mountain areas. Guidelines on how the farmers should deal with or prevent soil erosion are provided through the agricultural advisory service. Violating these guidelines can result in the reduction of direct payments. The next presentation, by T. Ziegler of the agricultural advisory service of the Canton Uri, questions the efficiency of subsidies. He thinks that the system should be used in a more targeted way. But this would require a clear definition of what is desired from alpine agriculture: high productivity or preservation of the cultural landscape and biodiversity. Following this introduction, the case study on Urseren by the Environmental Geoscience Group of the University of Basel was presented to the audience.

In the afternoon, G. Tognina, of the Agency for Nature and Environment of the Canton Grisons, presented a methodology on how to document and map damages caused by extreme events in the Canton Grisons. According to his experience, the monitoring proved very efficient in order to restore the affected areas and to assign compensation payments. Extreme events are responsible for the highest erosion yields. During a single event, hundreds of landslides may occur, as happened in August 2005. C. Rickli, from the Swiss Federal Institute for Forest, Snow and Landscape Research, showed the data from a large-scale assessment of landslides and related causal factors. Clearly, landslide occurrence is governed by topographic conditions, but differences between different types of vegetation cover and land use were also observed. Forested areas destabilize at higher slope angles. Moreover, the condition of the forest had a strong effect. Regarding grasslands, the highest landslide density was observed on intensively used cattle pastures followed by abandonment of the site. As such, the data support the findings of E. Tasser.

But what should be done when landslide damage has occurred? For instance the Canton Nidwalden was heavily affected by landslides after the August 2005 event. About 800 slides with a volume between 1000 and 100 000 m³ occurred. G. Richner from the Agency for Environment of the Canton Nidwalden reported on the restoration of landslide-affected grasslands. Several technical solutions are available for stabilizing a slope, for example, though wood or stone constructions. Besides these measures, drainage of the area and revegetation are important. The latter is a difficult task since the fertile subsoil layer is usually eroded. As such the stabilized and restored slopes might be of limited use for the farmer over longer periods.

G. Schmidt from the Environmental Protection Agency of the Canton St. Gallen and R. Sutter from the engineering office Agricultura in Appenzell presented the project “Erosion im Alpgebiet”, which was funded by the FABO of the cantons Appenzell Innerrhoden, Glarus, and St Gallen. The outcome of this work was the flyer “Alpenbewirtschaftung und Bodenerosion” published by the Swiss Association for the Development of Agriculture and Rural Areas (AGRIDEA). It is a tool to help farmers and agricultural advisers to realize where soil erosion risk may occur, which measures are needed to prevent soil erosion, and how damages can be restored and to distinguish unavoidable from avoidable soil erosion, which is the most difficult task. The study is based on investigation of 15 alps in the three cantons and the resulting recommendations for land management are basically congruent with the ones presented in chapter 6.

Erosion, loss of organic matter, compaction, salinization, landslides, contamination, and sealing, which are embraced by the term “soil degradation”, are accelerating, with negative effects on human health, natural ecosystems, and climate change as well as on our economy. At the moment, only nine EU Member States have specific legislation on soil protection (especially on contamination). Different EU policies (for instance on water, waste, chemicals, industrial pollution prevention, nature protection, pesticides, agriculture) are contributing to soil protection. But as these policies have other aims and other scopes of action, they are not sufficient to ensure an adequate level of protection for all soils in Europe.

For all these reasons, the Commission adopted a Soil Thematic Strategy (COM [2006] 231) and a proposal for a Soil Framework Directive (COM [2006] 232) on 22 September 2006 with the objective of protecting soils across the EU. L. Montanarella from the European Commission, Joint Research Centre (JRC), Italy, identified the following obstacles to implementation of soil erosion conservation in the Alps: i) there is no harmonized soil information system for the Alps; ii) a common high resolution digital terrain model is not available for research purposes in the Alps; iii) there is a lack of validated soil erosion models for Alpine conditions; iv) field measurements of erosion are still scarce and are not using harmonized observation methodologies.

6.2 Workshop in Basel, 2012

Soil erosion in the Alps is a well-recognized problem and is identified as a priority for action within the EU soil protocol of the Alpine Convention. There are several approaches to estimating soil erosion rates ranging from long-term quantification with cosmogenic nuclides to capturing short-term processes with modelling, remote sensing tools, and monitoring or empirical determinations using radiogenic isotopes such as caesium or beryllium or even stable isotopes ($\delta^{13}C$). Finally, soil erosion can be quantified from the off-site perspective using sediment flux measurements in streams and lakes. The workshop in 2012 aimed to bring these different scientific approaches together. This workshop in Basel followed the tradition of the first workshop, held in Andermatt, Canton Uri, Switzerland, in 2007, in bringing together scientists and stakeholders to discuss scientific results and perspectives on soil erosion in the Alps. In the 2012 workshop we addressed a broader perspective considering other alpine states and international views.

Views, perspectives, and thoughts from stakeholders

Roland von Arx of the Federal Office for the Environment pointed to the importance of soil protection knowledge and implementation of good management practice in the Swiss Alps in general and of soil erosion assessment more specifically. He formulated the hope that the workshop will improve the transfer of knowledge on Alpine soil erosion processes between the different Alpine states and also deepen the understanding of processes due to the participation of a broad range of different disciplines.

C. Wirz from the Federal Office for Spatial Development (ARE) raised awareness of the high soil consumption rates in Switzerland, which might increase the general importance of Alpine land and Alpine soils. With the loss of nearly 0.9 m² of soil every second to construction or sealing, most of it being highly productive, high quality agricultural soil, Alpine soils might become a valuable resource in the near future.

A. Candinas from the Federal Office for Agriculture (FOAG), Switzerland, initially stated that “agricultural policy has the objective to improve the actual situation”. In addition to existing environmental protection legislation, soil erosion prevention in Switzerland is also regulated via the direct payment ordinance. The proof of ecological performance (PEP), which is binding for farmers wishing to receive direct payments (98% participation), includes regular crop rotation and suitable soil conservation. However, so far the implementation of the legal requirements varies widely among different cantons.

To improve the situation, FOAG mandated the University of Bern to develop a soil erosion risk map of Switzerland in cooperation with the Agroscope Reckenholz-Tänikon research station. This work has been finalized and a precise high-resolution risk map (2 × 2 m grid width) is now available. As a first step towards implementation, this map has been published on the Website of FOAG. The map shows that around one third of the arable land has a high potential risk of soil erosion. Every farmer can exactly verify how his or her fields are classified in the erosion risk map and check whether the current cultivation is appropriate. All these measures are appropriate for arable land. However, two thirds of the agricultural area in Switzerland is permanent grassland, where soil erosion also occurs. Surely the economic importance of the grassland area is small in comparison with the arable area. Nevertheless, the grassland area is of very high importance for the future of agriculture and tourism in Switzerland. Thus it would be very useful for the FOAG to have a map of soil erosion risks in the grassland area of the same quality as the map for arable land.

A. Imhof (Environmental Protection Agency, Canton Uri, Switzerland) presented his view as an executing cantonal authority dealing with soil erosion in the Alps. In the Canton Uri, where he is responsible for quality soil protection (among other tasks), there is practically no farming other than pastures and meadows. Intensive cultures are more or less non-existent. Two-thirds of the utilizable space is alpine pastures. Every year, almost 10 000 cows, cattle, and calves, 17 000 sheep, and 1500 goats spend the summer months in the Alps of Uri. According to A. Imhof, soil erosion due to trampling is of minor importance. Only at the end of autumn when the grounds are wet may some damages occur. An alpine farm in Uri normally has two or more pastures at different altitudes. The lowest one is used in early summer and autumn, the highest during midsummer. Thus, the meadows have time to recover. Traditional alpine farming ensures regeneration and therefore helps to prevent erosion. According to A. Imhof the question of implementation needs more attention. Scientifically based and easily applied erosion assessment criteria are needed. The setup of legal and administrative guidelines treating soil protection and agriculture in the cantons by the Federation was a great step forward. These guidelines help to assess a situation on-site and to decide what to do. However, it remains unknown whether the cantons will provide the required money and instruments for implementation. A. Imhof pointed out that the flyer published by AGRIDEA about soil erosion in alpine pastures is an example of a very helpful tool for the implementation, because it shows the farmers how to easily prevent soil erosion. From scientists he would expect easy and reliable methods to assess soil erosion, practical measures against soil erosion, and finally regular consultation with the people directly involved.

P. Panagos presented the talk of L. Montanarella from the European Commission, JRC, Italy. With regard to Europe, it has been found that almost 12% of the European territory, which amounts to almost 115 million hectares, is subject to soil erosion. The impact of soil erosion in terms of financial cost has been calculated to be several billion euros. The introduction of the Thematic Strategy for the Protection of Soils and the establishment of the European Soil Data Centre (ESDAC) at the European Commission are recent developments in soil policy at the European level. ESDAC provides a mechanism for reporting soil critical data by EU Member States with the ultimate goal of assessing soil conditions from harmonized data sources. In order to develop a European-wide reference dataset for soil erosion, ESDAC has made use of the European Environment Information and Observation Network for Soil (EIONET-SOIL), whose members (the National Reference Centres for Soil) were asked to provide their best data on actual soil erosion.

By 2020, EU policies will take into account their direct and indirect impact on land use in the EU and globally. In addition, the rate of land take is on track to meet the aim of achieving no net land take by 2050; soil erosion has been reduced and the soil organic matter has increased, with remedial work on contaminated sites well underway. The specific targets to be met by 2020 are as follows: (i) ecosystems and their services should be maintained and enhanced by establishing green infrastructure and restoring at least 15% of degraded ecosystems, (ii) annual land take (i.e. the increase of artificial land) should not exceed 800 km^2 per year at EU level, (iii) the area of land in the EU that is subject to soil erosion of more than 10 tonnes per hectare per year should be reduced by at least 25%, iv) soil organic matter levels should not decrease overall and should increase for soils that currently have less than 3.5% organic matter (soil organic carbon 2%).

The European Commission organized a conference on “Land and Soil Degradation Post Rio+20” in Brussels in November 2012. The objective of these events was to discuss the European answer to the land and soil commitments made by world leaders at Rio+20. In the follow-up of Rio+20, “The Future We Want”, the long-term goal is zero net land degradation.

The role of snow and winter processes in soil erosion

The different types of avalanches and their characteristics were introduced by M. Freppaz from the University of Turin, Italy. Most frequently, avalanches occur at slope angles between 30 and 45°. While dry snow avalanches hardly affect the soil surface, wet snow avalanches, especially of the glide avalanche type, occur at the ground (soil)–snow interface and are erosive. Three conditions need to be fulfilled to form glide cracks prior to the avalanche release: (i) a snowpack–ground (soil) interface of low roughness, (ii) a temperature equal to 0 °C at the snow–ground (soil) interface, which allows for the presence of free water, and (iii) a slope angle >15°. The avalanches are particularly erosive if the entire snow depth is involved and the snow runs onto snow-free areas. An already developed 1-D avalanche dynamics model was modified to include soil erosion. The key features of this model are (i) the estimation of the avalanche ground shear and (ii) the estimation of the soil critical shear stress. The simulation shows that avalanche ground shear is 1.2–1.5 kPa and thereby exceeds the soil critical shear stress, which is <10 Pa, more than 100 times. For a study site with frequent avalanche release, he reported an average mass removal of between 0.07 and 0.8 t ha^{-1} $event^{-1}$.

E. Ceaglio of the University of Turin, Italy, reported high erosion rates of 3.7 and 20.8 t ha^{-1} triggered by wet avalanche release in the Aosta Valley (Italy). The second event even exceeded the long-term annual soil losses of 13.4 t ha^{-1} yr^{-1} in the avalanche release area and 8.8 t ha^{-1} yr^{-1} in the track area. The results confirm that in areas frequently affected by wet avalanches, this is the main process of soil loss. The following presentation on the effect of snow gliding on soil erosion by G. Leitinger from the University of Innsbruck, Austria, was cancelled.

Landslide mapping and prediction

J.-P. Malet, from the University of Strassbourgh, France, focussed on the effects of climate change on the activity of landslides with examples from the south French Alps. The evaluation of past landslide triggering revealed that(i) events characterized by high rainfall intensity and short episode duration (i.e. mostly the result of localized convective storms) will trigger mostly debris flows and shallow slides in relatively permeable soils (e.g. moraines, scree slopes, or poorly sorted slope deposits), and (ii) long rainfall periods characterized by low to moderate average and peak rainfall intensity (i.e. the result of multiple and successive storms during a period of several weeks or months) can trigger or reactivate shallow and deep-seated landslides in low permeability soils and rocks (e.g., black marls, clay-rich material). A process-based model of slope hydrology/stability was used to assess the impacts of climate change on landslide frequency. The results of the simulations show that landslides triggered by a high groundwater level are likely to decrease with the changed climate due to the general decrease in groundwater levels and amounts of water storage in the soils.

C. Wiegand, from the Austrian Academy of Sciences, Innsbruck, Austria, presented a talk about automated extraction of shallow erosion areas based on multi-temporal ortho-imagery. The method was tested in the inner Schmirn Valley, Austria. The presentation covered both the detection of erosion areas and a multi-temporal analysis of the geomorphological changes. The presented approach is an appropriate tool for detecting shallow landslides. The multi-temporal analysis revealed a high dynamic of the observed shallow landslides. The rates of increase between the observation years are irregular and most likely related to the occurrence of single triggering events. In contrast, the rates of decrease are almost constant and suggest a continuous rate of recovery.

Causes and risk factors for soil erosion

Nicolle Mathys (Centre National du Machinisme Agricole, du Genie Rural, des Eaux et des Forets [CEMAGREF], Grenoble, France) reviewed the results of 27 years of monitoring in the Black Marl formations of the French south Alps. The erosion rates of three small basins, at 59–132 t ha^{-1} yr^{-1}, are extremely high due to the special geologic and geomorphological setting but also due to intense rainfall events. However, there is a large seasonal effect on sediment supply which is not totally explained by the seasonality of the rainfall inputs. This could be explained by a seasonal model of sediment production. In winter, freezing–thawing cycles generate a thick mantle of weathered material. In spring, the first storms move the material accumulated in the rills and small mudflows occur on steep slopes. If the runoff in the drainage network is sufficiently high, the accumulated materials of the foot slopes are transported to the outlet. If not, they increase the stocks in the reaches. The intense summer storms, especially when they contain hail, move considerable amounts of sediments. Finally, in autumn, the lower intensities and the decrease in the availability of weathered material produce less erosion on the slopes.

In Austria there are indications of an increase in soil erosion (mainly shallow landslides) in treeless areas in the high montane and subalpine zone of the Alps. The reasons for this increase, which occurred in the second half of the twentieth century, are not clear and are thus the subject of a recently started interdisciplinary research project called EROSTAB. B. Kohl from the Federal Research and Training Centre for Forests, Natural Hazards and Landscape, Innsbruck, introduced the project to the audience. The project aims to understand which combination of factors is causing the increase in soil erosion in order to improve future predictions and early detection of instable slopes. Monitoring of soil erosion features will be done with aerial photos and airborne laser scanning datasets. The erosion-relevant parameters surveyed in the field will be analysed through geo-statistics and modelling. The first results are congruent with the findings of the University of Basel and indicate that (i) a decline of pasturing leads to higher shallow landslide susceptibility and that (ii) very intensive mechanical impact of livestock and (iii) snow gliding and wet avalanches might have an impact.

C. Rixen of the Swiss Federal Institute for Forest, Snow and Landscape Research, Switzerland, focused on restoring an intact plant cover and effects of biodiversity. The effects of alpine plants on soil are diverse: (i) the number of plant species is positively correlated with soil aggregate stability, (ii) the percentage vegetation cover strongly determines soil erosion, and (iii) below-ground diversity of roots is crucial for slope stability. C. Rixen is moreover engaged in the Swiss working group for high altitude restoration and presented the approach and study site in the Bedretto Valley.

J. Troxler from Agroscope, Changin, Switzerland, reviewed the potential impacts of mountain agriculture on soil erosion and strategies for mitigation. This topic has been discussed in more detail in chapter 6.

V. Zupanc of the University of Ljubljana, Slovenia, reported on soil erosion in the Julian Alps in the Triglav National Park, which is the largest protected natural area in Slovenia. The evaluation of meteorological data for 18 stations reveals a significant increase of annual average air temperature. In a selected case study, eroded material considerably decreased the pasture area, which resulted in a reduction of subsidies for the land owner.

N. Kuhn of the University of Basel, Switzerland, presented data on selective erosion and transport of aggregates. He pointed to the importance of soil erosion and the related movement of soil organic matter.

Methods to quantify soil erosion

L. Mabit (International Atomic Energy Agency Vienna, Austria) introduced the concept of fallout radionuclides for soil erosion assessment. Even though the classical approach of using radionuclides for soil erosion assessment has been successfully applied for large areas (~200 km²), the application in the Alps should be restricted to areas in the vicinity of the reference sites. In the Alps the underlying assumption of the ^{137}Cs method that the fallout occurred homogenously is not met since the Chernobyl-^{137}Cs contribution is relatively high. The latter fallout occurred mainly during a few events after the accident and is thus affected by heterogeneous rainfall distribution patterns. Moreover, during the time of the fallout, some elevated parts were still snow covered, which caused heterogeneous input of ^{137}Cs during the subsequent snowmelt. The problem heterogeneous ^{137}Cs distribution can be tackled though resampling of ^{137}Cs and through the use of $^{239+240}Pu$, as presented by C. Alewell (more details have been discussed in chapter 3.1.2).

Still, radionuclides are useful for large areas if their fingerprinting information is combined with suspended sediment monitoring, as presented by O. Evrard of the Laboratoire des Sciences du Climat et de l'Environnement (LSCE/IPSL), France. A successful application was done in the sub-alpine Bléone catchment (southeast France). Between 2007 and 2009, erosion rates reached 2.49 ±0.75 t ha^{-1} yr^{-1} at the outlet of the Bléone catchment, but this mean value masked important spatial variations of erosion intensity within the catchment of 0.85–500 t ha^{-1} yr^{-1}. Quantifying the contribution of different potential sources revealed that even though Black Marls generated locally very high erosion rates (as also presented by N. Mathys), they supplied only a minor fraction (5–20%) of the fine sediment collected on the riverbed in the vicinity of the 907 km² catchment outlet. The bulk of sediment originated from Quaternary deposits (21–66%), conglomerates (3–44%), and limestones (9–27%).

Challenges in modelling of soil erosion on large scales

The greatest obstacle to soil erosion modelling at larger spatial scales is the lack of available soil data as stated by P. Panagos from the JRC, Italy. A key parameter for modelling soil erosion is soil erodibility, expressed as the K-factor in the commonly used soil erosion model, the USLE, and its revised version, RUSLE. The K-factor, which expresses the susceptibility of a soil to erosion, is related to soil properties that trigger erosion such as organic matter content, soil texture, soil structure, and permeability. Since the Land Use / Cover Area frame Survey (LUCAS) soil survey in 2009, a pan-European soil dataset is available for the first time, consisting of around 20 000 points across 25 Member States of the European Union. The soil erodibility dataset overcomes the problems of limited data availability for K-factor assessment and presents a high quality resource for modellers who aim at soil erosion estimation on local/regional, national, or European scale.

A second critical point concerns the validation of such maps. In 2010, the European Soil Data Centre conducted a project to collect data on soil organic carbon and soil erosion from national institutions in Europe using the European Environment Information and Observation Network for soil (EIONET-SOIL. So far, complete soil erosion datasets have been received from national institutions for only eight countries and are compared to the pan-European model estimates done with RUSLE and PESERA.

Regarding soil erosion modelling for the Alps, K. Meusburger stated that Switzerland might play a key role since topographic and meteorological data such as rainfall erosivity are available in good quality.

Soil erosion and soil formation from a geomorphological perspective

M. Keiler of the University of Bern reported on the influence of climate change on hazard processes in the Alps. The European Alps are disproportionately affected by ongoing climate change, which is superimposed upon a longer-term paraglacial signal that corresponds to processes of landscape readjustment/relaxation following the last glacial event. The deglaciated, steep, scarcely vegetated and often water-saturated slopes have a natural disposition for natural hazards such as rock falls, landslides, and debris flows. As such, alpine areas are in transition from a glacial to a non-glacial state. Two important characteristics of this transition period are glacier retreat and melting of alpine permafrost, both of which are known from field studies across the European Alps. At higher elevations, these processes are genetically associated with land-surface instability and enhanced sediment delivery to mountain slopes and valley bottoms. Consequently, it is likely that ongoing and accelerating ice loss in the European Alps over the next decades to centuries will have significant impacts on hazard type, location, and frequency. As the understanding of future climate impacts is hampered by problems of general circulation models downscaling in areas of complex local relief and microclimate, future research on geomorphic processes and monitoring of land-surface systems is needed to establish the sensitivity of these systems to climate forcing. This knowledge is crucial to improve hazard and risk management in the European Alps.

The geomorphologists M. Egli (University of Zürich), F. Schlunegger (University of Bern), and F. Kober (ETH Zürich) presented long-term denudation rates and their influencing factors for the Alps. M. Egli defined denudation (D) as the sum of physical erosion fluxes (E) and chemical weathering fluxes (W), with D = E + W. Denudation rates are assessed with cosmogenic nuclides such as ^{10}Be. A steady state is observed if the denudation rate is equal to the soil and saprolith production.

A compilation by Salcher et al. (in preparation) showed that denudation rates vary from 0.16 to >1.5 mm yr^{-1} in the European Alps. Assuming a bulk density of the soil of 1 g cm^{-1}, this would correspond to 1.6 to 15 t ha^{-1} yr^{-1}. The denudation rates increase with increasing slope, elevation, geodetic uplift rate, and drainage density.

Soil formation (which is the net rate resulting from soil production minus soil denudation) rates in alpine areas vary with parent material and soil age. The younger the soil, the higher the formation rate. For a soil on siliceous material with an age >1000 years, the formation rate varies between 1 and 30 t ha^{-1} yr^{-1}, while for older soils the rate is below 1 t ha^{-1} yr^{-1}. On calcareous parent material, soil formation strongly depends on soil biology and water fluxes; values range between 0.1 and 0.5 t ha^{-1} yr^{-1}. In comparison, the Swiss Ordinance for Soil Protection provides guideline values for soil erosion of 2 to 4 t ha^{-1} yr^{-1} for shallow and deep soils, respectively. Even though soil formation in the Alps is not slow, this corresponds to the maximum formation rate of a young soil (<1000 yr). M. Egli concluded that the “Swiss are generous” with the loss of the resource soil.

7 > Conclusion

Soil loss in Alpine grasslands is mostly triggered by either sheet erosion or landslides. The natural disposition to sheet erosion and landslides is rather high in Switzerland due to the high rainfall erosivity and the steep slopes. However, the presented case studies on soil erosion in the Alps revealed that the anthropogenic impact is substantial. In fact, an increase of shallow landslides incidence of between 43 and 92% was observed in the three investigated catchments (Urseren, Obergoms, and Bedretto/Piora valleys) since the middle of the last century and is due to climate and land-use changes. The latter was also reported for the Austrian Alps.

Natural landslide susceptibility is mainly determined by slope angle, slope aspect, geology, and hydrological parameters. The methods of mapping landslides and landslide susceptibility are well known and used worldwide. However, it was shown that dynamic landslide causal factors such as climate (changes in rainfall and snow patterns) and land use can trigger an increase in land slide occurrence and even change the spatial patterns of landslide susceptibility. Our assessments show an increasing trend of landslide incidence for very intensively used sites, which is congruent with the findings of other studies. However, there is also evidence of increased landslide incidence at high altitude sites in the Bedretto/Piora and Obergoms valleys that have never been subject to land use. The most likely explanation for this observation is a change in snow patterns and snow dynamics due to climate change. Under the term "shallow landslides", different triggering processes are involved, for example triggering through rainfall, snowmelt, or abrasion by avalanches, snow glide, and creep. A distinction between these process types is not possible from aerial photograph mapping and is even difficult in the field. A clarification of the processes involved is further hampered by the lack of hydro(geo)logical, substrate, and soil data and by the limited understanding of the interaction between soil, plants, and snow. More detailed studies could improve our understanding of these processes and their prediction. We could not confirm the increased landslide susceptibility for abandoned sites that has been reported by other studies. The causal relation between abandonment and increased landslide susceptibility needs to be assessed carefully, since sites with high natural disposition are preferentially abandoned. Moreover, with the abandonment, the restoration of the site by the farmer is also given up, so damages are more persistent. Succession starts and alters, for example, the root density in the soil and the resistance of the plants to snow movement. These succession states are more susceptible to shallow landslides until larger shrubs and small trees can establish themselves.

The observed trends in landslide incidence cause a loss in prediction quality for future events. A multi-temporal susceptibility assessment, as presented for the Urseren Valley, can resolve the problem if sufficient land-use information is available. Soil loss by landslides varies strongly by location and event and is usually not assessed. For the Urseren Valley, which is heavily affected by shallow landslides, a rough estimate suggests that an average rate of 0.6 t ha^{-1} yr^{-1} is not exceeded if the whole catchment is considered. In comparison, the catchment-averaged soil erosion rate (sheet erosion) adds up to three times this amount (1.18 t ha^{-1} yr^{-1} in the case of the Urseren Valley). It would be desirable to integrate the process of shallow landsliding into soil erosion models for mountain areas in the future, to account for this additional soil loss.

Regarding sheet erosion, we showed that rainfall and especially vegetation cover are two main drivers. At the same time, these are the parameters most likely to be affected by climate and land-use changes; for example, clear increasing erosivity trends were observed for the months of May to October in Switzerland. The increasing trends of rainfall erosivity in May, September, and October, when vegetation cover is scarce, are likely to enhance soil erosion risk for certain agricultural crops and alpine grasslands in Switzerland. For detailed soil erosion risk assessment, a Swiss-wide mapping of monthly rainfall erosivities needs to be combined with the risk of soil being exposed by damaged vegetation cover. Furthermore, land-use management information needs to be considered. These two parameters are and will be of major concern for soil conservation. Even a simple combination of fractional vegetation cover mapping and rainfall erosivity per month would already provide a good estimate of the erosion risk in alpine areas.

All of the studies consulted indicate the importance of an intact vegetation cover in order to prevent soil erosion in alpine areas. For sites with reduced vegetation cover, erosion rates above 20 t $ha^{-1}yr^{-1}$ can be observed. In areas with erodible substrate like the Black Marls in the French Alps, even soil erosion rates over 50 t ha^{-1} yr^{-1} are possible (Evrard et al. 2011). Moreover, the vegetation cover also decisively influences the occurrence of avalanches and snow gliding. The investigations in the Aosta and Urseren valleys show that snow-induced soil erosion, with maximum values of 20.8 and 23.6 t ha^{-1} $winter^{-1}$, is of the same order of magnitude as water soil erosion during the summer season. These “winter erosion” processes and their spatial relevance need to be investigated in more depth with the long-term goal of including the process in soil erosion risk models. Recently, the first physically based attempts to model the erosive force of wet avalanches were made (Confortola et al. 2012). And the application of the empirical spatial snow-glide model (Leitinger et al. 2008) for the Urseren Valley indicates that a large part of the area is affected by very high potential snow-glide rates (Meusburger et al. 2013a).

In general, modelling the soil erosion risk of sheet erosion in alpine areas will remain a challenge but the presented investigations showed that:

> i) the focus should be on simple models such as RUSLE that are flexible and adaptable to most environmental conditions; for instance the erosivity of snow and the enhanced erodibility through trampling could be introduced through the R- and K-factors, respectively;
> ii) the use of high quality input data is also very promising with respect to validation;
> iii) validation of the model estimates by field measurements is crucial.

A large part of the research of the University of Basel was dedicated to the evaluation and development of appropriate methods for soil erosion identification and quantification in the field. The classical methods like sediment traps or mapping of erosion features are only partly suitable. However, isotopes have proved to be suitable to identify soil disturbance and to quantify soil erosion rates. The latter is possible through the use of the radionuclides ^{137}Cs and $^{239+240}Pu$.

Soil erosion and landslides are and will always be inevitable processes in alpine areas since these areas have the highest potential soil erosion risk in Europe. The actual soil erosion rate can only be significantly reduced through careful land-use management that aims at the preservation of an intact vegetation cover. Strategies for careful grassland management are well known but need to be adapted and implemented, depending on the natural disposition of the site. Therefore a spatial assessment of potential soil erosion risk in alpine areas is needed to adapt and optimize the land-use management.

Finally, to assess sustainable soil use and management in alpine areas, rates of soil production need to be considered and balanced with soil loss rates. Soil production is a function of time and ranges from approximately 0.5 $t\ ha^{-1}\ yr^{-1}$ to a maximum rate for young soils of 9 $t\ ha^{-1}\ yr^{-1}$ (for siliceous parent material; (Alewell et al. 2014). Consequently, for sustainable land use, erosion rates should be equal to or lower than these values. Soil erosion assessments all over the Alps showed that this rate is frequently exceeded and thus sustainability is not ensured.

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> Abbreviations, figures and tables

Figures

Tables