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Modeling soil erosion and reservoir sedimentation at hillslope and catchment scale in semi-arid Burkina Faso

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ABSTRACT

Soil erosion is a major factor of land degradation in Sub-Saharan Africa. The loss of nutrient-rich topsoil from hillslopes causes severe agricultural problems for an extremely vulnerable agricultural society that depends on soil quality as a fundamental base for its livelihood. The removal of soil in source areas leads to sediment accumulation in sink areas such as dammed reservoirs. Especially the siltation of small reservoirs is seen as a serious environmental threat in Burkina Faso, where more than a thousand dams have been built to store unevenly distributed rainwater for the dry season. These dams are in danger of losing their function as essential water reservoirs for domestic use, irrigation and stock watering in the near future. This study presents an integrative, scale-dependent approach to assess on-site and off-site impacts of soil erosion by quantifying the magnitude and intensity of soil loss/deposition at hillslope and catchment scales and by considering the spatial dimension of these processes in a complex landscape system in southwestern Burkina Faso.

At the hillslope scale, the spatial variability of soils is analyzed by soil profile investigations along a catena and subsequently considered in soil erosion simulations by the physically-based WEPP model. WEPP model predictions indicate that although average soil loss rates simulated for the entire hillslope are comparatively low, they can be forty times higher at particular hillslope positions. These spatial differences, even in the relatively flat terrain of Burkina Faso, are also confirmed by ^{137}Cs measurements with averaged soil loss rates of less than 5 t ha^{-1} yr^{-1} and maximum erosion rates of more than 50 t ha^{-1} yr^{-1} in erosion hotspots. The identification of such hazard zones can be used to target site-specific land management options. WEPP model simulations show that the application of stone lines, minimum tillage, contour farming and residue management could reduce soil loss by up to 95 %, 70 %, 55 % and 45 % at these erosion-prone hillslope positions.

At the catchment scale, sedimentation rates of three reservoirs are analyzed by bathymetric surveys, sediment core sampling and sediment yield calculations using the soil erosion and sediment delivery model WaTEM/SEDEM. For the model, a digital elevation models is generated and land-cover maps are derived from remote sensing images. A comparison between the initial and actual reservoir bed morphology shows that the reservoirs have lost approximately 10-15 % of their original storage capacity and more than 60 % of their inactive storage volume in the last 15 to 20 years. During that period, a sedimentation layer of 0.3 m to 0.5 m thickness has accumulated on the reservoir bed. This was verified by stratigraphical changes and downcore variations in sediment properties and ^{137}Cs concentrations. Predictions by WaTEM/SEDEM show similar magnitudes of siltation with specific sediment yield rates of 0.5 t ha^{-1} yr^{-1} to 3.4 t ha^{-1} yr^{-1}.

These results indicate that the half-life of the dams might be reached in about 25 years assuming constant siltation rates under current conditions. In order to identify the sediment source areas and the potential soil-erosion risk zones leading to these high siltation rates, a spatially-explicit soil erosion and deposition hazard maps generated by WaTEM/SEDEM can be used. These hazard maps present a powerful tool to support policy makers in their decisions on which landscapes are primarily at risk and where action plans for sustainable soil and water conservation should be implemented.

Modellierung von Bodenerosion und Sedimentation von Stauseen auf Hang- und Wassereinzugsgebietsebene im semi-ariden Burkina Faso

KURZFASSUNG

Die Bodenerosion hat einen wesentlichen Einfluss auf die Landdegradierung semi-arider Gebiete in Afrika südlich der Sahara. Der Abtrag von humusreichem Oberboden am Hang verursacht schwerwiegende landwirtschaftliche Probleme, insbesondere für eine fragile Ackerbaugesellschaft, die von einer guten Bodenqualität zur Sicherung ihrer Existenzgrundlage abhängig ist. Der Abtrag von Bodensedimenten aus Quellgebieten hat gleichzeitig die Akkumulation von Sedimenten in Zielgebieten wie beispielsweise eingedämmten Rückhaltebecken zur Folge. Insbesondere die Verschlämmung von Kleinstauseen stellt für das Land Burkina Faso, in dem über tausend Staudämme gebaut worden sind, um das Regenwasser der uneinheitlich verteilten Niederschläge für die Trockenzeit stauen zu können, ein zunehmendes Umweltproblem dar. In naher Zukunft drohen diese Staudämme ihre Funktion als unverzichtbare Wasserspeicher für Haushalt, Bewässerungsfeldbau und Viehzucht zu verlieren. Um die Auswirkungen von Bodenerosion sowohl on-site als auch off-site zu bewerten, verfolgt die vorliegende Arbeit einen integrativen, skalenabhängigen Ansatz, bei dem einerseits das Ausmaß und die Intensität von Bodenerosion und deren Ablagerung auf Hang- und Einzugsgebietsebene quantifiziert werden und bei der andererseits die räumliche Dimension dieser Prozesse in einem komplexen Landschaftssystem im Südwesten Burkina Fasos berücksichtigt wird.

Auf der Hangebene wird die räumliche Variabilität von Boden anhand von Bodenprofilen entlang einer Catena untersucht und in die Erosionsmodellierung mittels des physikalisch-basierten WEPP-Model miteinbezogen. Die Ergebnisse der Modellierung verdeutlichen, dass, obwohl die simulierten Abtragsraten für den Gesamthang als vergleichsweise gering einzuschätzen sind, diese bis zu vierzigfach erhöht an einzelnen Hangbereichen auftreten können. Auch ^{137}Cs Messungen bestätigen mit durchschnittlichen Abtragsraten von weniger als 5 t ha^{-1} yr^{-1} für den Gesamthang und maximalen Abtragsraten von mehr als 50 t ha^{-1} yr^{-1} in gefährdeten Zonen diese hohen, räumlichen und für die verhältnismäßig flache Landschaft Burkina Fasos auffälligen Differenzen. Die Identifikation dieser Gefährdungszonen kann jedoch einer standortspezifischen Anpassung von Landnutzungsmethoden dienen. Die Simulationsergebnisse des WEPP-Models zeigen, dass durch die Anlegung von Steinwällen, weniger intensiven Bodenbearbeitungsmethoden, Konturpflügen und Mulchsaat der Bodenabtrag an diesen besonders erosionsgefährdeten Hangbereichen um bis zu 95 %, 70 %, 55 % und 45 % reduziert werden könnte.

Auf Wassereinzugsgebietsebene wird die Sedimentationsrate von drei Kleinstaudämmen durch bathymetrische Methoden, Sedimentbohrungen und die Berechnung der Sedimentfracht anhand des Bodenerosions- und Sedimentaustragsmodelles WaTEM/SEDEM ermittelt. Für die Modellierung wird ein digitales Höhenmodel erstellt, Landbedeckungskarten werden wiederum von Fernerkundungsbildern abgeleitet. Ein Vergleich zwischen der ursprünglichen und aktuellen Stauseemorphologie zeigt, dass die Staubecken in den letzten 15 bis 20 Jahren zwischen 10-15 % ihrer gesamten Speicherkapazität und mehr als 60 % ihres inaktiven Speichervolums eingebüßt haben. In diesem Zeitraum hat sich auch eine Sedimentschicht von 0.3 m bis 0.5 m Mächtigkeit auf dem Stauseeboden abgelagert,

was mittels stratigraphischer Analysen und Veränderungen sedimentspezifischer Eigenschaften sowie ^{137}Cs Konzentrationen im Bohrkern belegt wird. Modellberechnungen mit WaTEM/SEDEM bestätigen einen ähnlich hohen Verschlämmungsgrad mit spezifischen Sedimentfrachtraten von 0.5 t ha^{-1} yr^{-1} bis 3.4 t ha^{-1} yr^{-1}.

Die Ergebnisse verdeutlichen, dass - setzt man gleichbleibende Sedimentationsraten unter den heutigen Bedingungen voraus - die Halbzeit der prognostizierten Lebensdauer der Staudämme in etwa 25 Jahren erreicht sein wird. Um Sedimentbereitstellungsgebiete und potentielle Erosionsrisikobereiche, die zu diesem hohen Sedimenteintrag in Stauseen führen, zu identifizieren, dienen räumlich-detaillierte Bodenabtrags- und Akkumulationsgefahrenkarten, die mit WaTEM/SEDEM erstellt werden. Diese Gefahrenkarten können schließlich von Entscheidungsträgern als sinnvolle Planungsgrundlage genutzt werden, um festzulegen, welche Landschaftsbereiche vorrangig als Risikogebiete ausgewiesen werden sollten, um dort nachhaltige Boden- und Wasserschutzmaßnahmen zu implementieren.

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1 INTRODUCTION

1.1 Problem statement

Soil degradation is recognized as a serious environmental threat in Sub-Saharan Africa (Aubert, 1948; Boyer, 1963; Lal, 2001; Roose, 1994; Symeonakis and Drake, 2010; Vlek et al., 2008), not only due to a highly vulnerable soil-climate-ecosystem, but also because of a extremely vulnerable agricultural society depending on soil quality as a fundamental base for its livelihood. Soil erosion is one of the most common and widely spread forms of soil degradation and is closely related to processes of nutrient depletion and desertification in semi-arid environments (Bationo et al., 2007; Cobo et al., 2010; Koning and Smaling, 2005; Richard, 1990; Roose and De Noni, 2004; Sivakumar, 2007). The loss of nutrient-rich topsoil through erosion may lead, moreover, to severe changes in soil nutrient status and soil structure characteristics, which can pose considerable restrictions for agricultural use due to diminished soil fertility or the formation of hardpans (Brown et al., 1994; Leprun, 1981; Stoorvogel and Smaling, 1998; Pierce and Lal, 1994; Vanlauwe, 2006). Additionally, socio-economic developments, demographic dynamics and political interventions can lead to changes in people’s environmental needs and often present new challenges for agricultural societies in terms of land-use strategies and natural resources management (Batterbury and Warren, 2001; de Jager, 2005; Fairhead and Scoones, 2005; Heerink, 2005; Smaling and Dixon, 2006; Warren, 2002).

In Burkina Faso, ample attention has been paid to on-site effects of soil erosion, directly affecting farmers as erosion occurs on their fields. Farmers in an agriculture-dominated society - the primary sector in Burkina Faso employs 85% of the population and comprises approximately 70% of all export products (SP/CONAGESE, 2000) - are highly dependent on their soil resources to ensure adequate crop yield to maintain their livelihoods. This is often difficult in a country that belongs to both the economically poorest and most resource-limited countries in the world (HDI rank 177 out of 182; UN Human Development Report, 2009) and that is characterized by erratic rainfall and water shortage due to its semi-arid climate. In recent years, many studies have emerged that consider soil erosion in the context of soil conservation techniques (Kiepe et al., 2001a; Reij and Thiombiano, 2003; Roose et al., 1999; Sidebé, 2005; Zougmoré et al., 2004a), nutrient status (Visser and Sterk, 2007), sustainable livelihoods (Hessel et al., 2009; Sedogo, 2002), and indigenous knowledge systems (Gray and Morant, 2003; Mulders, 2001; Niemeijer and Mazzucato, 2003).

At the policy level, soil erosion has been identified as a major cause of the physical degradation of land. The Panafrican Conference, held in March 1997 in Ouagadougou convened by the UN Convention to Combat Desertification (UNCDD), called for action to solve the rapid formation of hardpans, which limit infiltration and increase runoff. Since then, there have been many political interventions on regional, national and international levels (e.g., CCD, 2004; UNDP, 1999; UN-Millenium Project 2005), and the number of politically driven, problem-oriented and applied interventions has risen.

However, the soil erosion research community has paid less attention to the spatial dimension of erosion, and only a few studies consider both on-site and off-site effects of erosion (e.g., Boardman et al., 2009; de Vente, 2005; Tamene, 2006; Verstraeten et al., 2003). Soil erosion is not only an in-situ phenomenon but also part of a complex sediment budget chain involving processes related to sediment transport. This is evident from rills or gullies in the landscape as well as sediment deposition, which includes all kinds of sediment storages, such as intermediate sinks downslope and long(er)-term storage in valleys, streams or reservoirs (Foster, 2006; Owens and Collins, 2006; Walling, 2006; Wilkinson et al., 2009).

Sedimentation, especially of small reservoirs, plays an important role in Burkina Faso (Cecchi et al., 2009). The country belongs to the large watershed of the Upper Volta Basin and is characterized by a high number of small dams, which were built to overcome water shortages during the dry season. The construction of small dams is a very common practice for dealing with erratic and unevenly distributed rainfall in semi-arid countries. Retained rainfall water in reservoirs or retention basins can ensure sufficient water availability for households, livestock and agriculture during the dry season. As many of the dams were built as a result of development aid programs in the 1980s, it is likely that the half-life of some of these reservoirs will soon be reached. This would imply that half of the original water storage capacity has been lost, with severe impacts on water availability and water use potential. Although it is controversially discussed whether dams are still an adequate option for dealing with water allocation conflicts in developing countries, problems related to existing dams should not be neglected as they will affect the local population, which still benefits from the multiple functions of reservoirs. On the contrary, more attention should be given to concerns about how to handle accumulated sediment and how to prevent, or at least slow down, the process of siltation by, for example, applying soil and water conservation techniques in sediment source areas.

The assessment of on-site and off-site effects requires not only an analysis of the magnitude and intensity of soil erosion/deposition processes but also of the spatial dimension of these processes in a complex landscape system, which inevitably leads to the question of scale. In addressing the scaling issue all environmental processes and landforms should be considered within the specific spatial and temporal dimension in which they operate (operational scale), in which they can be measured in the field (observational scale), and finally in which they can be captured for modeling (modeling scale) (Zhang et al., 2004). These phenomenological and methodological scales have recently gained greater significance in understanding the complexity of environmental systems followed by discussions about transferring knowledge between systems of different magnitudes, scaling relations in soil erosion and sediment budgets, and scale dependency of parameters for up- and downscaling (Beven, 1995; Blöschl and Sivapalan, 1995; Dikau, 2006; Kirkby, 2001; Slaymaker, 2006; Sivapalan et al., 2003; Summerfield, 2005). For soil erosion research, these considerations are essential, as the phenomenon of soil erosion involves natural processes and characteristics on multiple scales, such as soil surface and erodibility characteristics on a plot scale (including, e.g., aggregate stability, micro- and macropores or infiltration capacity), vertical and lateral flow characteristics on a hillslope scale (including, e.g., slope properties, rill and interrill development, runoff generation, overland flow or sediment routing), and finally dynamic interaction of soil and water movement on broader catchment or landscape scales (including, e.g., spatial variations in soil types, vegetation cover, relief or even rainfall intensities). Although, most of the processes are not limited to one specific scale and "each level of the hierarchy includes the cumulative effects of lower levels but also new considerations (emergent properties)" (Slaymaker, 2006, p.8), it is both effective and functional to identify the most dominant processes at each scale in order to select or adjust soil erosion measurements and modeling approaches (Kirkby, 1998).

Spatial- scale based soil erosion assessment requires a comparison of the magnitude of soil erosion at each scale by using individual, scale-dependent pedo-geomorphological methods and spatially explicit models. Moreover, knowledge about the spatial pattern of soil re-distribution on hillslope, catchment or landscape scale might be more meaningful than single numbers and averaged soil loss estimates. While single numbers on a regional or even national scale can be used to show the significance of soil erosion and emphasize the need for erosion control techniques (Oldeman et al., 1991), they might be misleading at smaller scales, as they cannot provide insight about the severity of erosion on smaller regional or catchment scales (Boardman, 2006). Such indicators might therefore not be suitable as a reference for local actors such as NGOs, stakeholders, extension workers or even farmers. If, however, small-scale erosion maps include information about the spatial pattern of soil redistribution rates, they could allow the identification of erosion-susceptible areas (so-called hot spots) more precisely and could additionally facilitate the selection of soil conservation and land management methods to fit site-specific landscape conditions.

Spatial scale in a complex landscape can be dealt with by delineating the area into different zones in which specific processes tend to be dominant. The catena concept presents such a theoretical framework on a hillslope scale, and ascribes systematic changes in soil properties to pedo-geomorphological and hydrological characteristics along the slope, influencing in return soil erosion susceptibility, sediment transport and soil redistribution patterns (Conacher and Dalrymple, 1977; Milne, 1935; Ruhe, 1960). The catenary soil succession is not just the end product of slope processes but reflects an interacting process between soil properties and landform, the so called pedo-geomorphology. A wide range of geomorphological studies in West Africa consider soil genesis, geological landscape formation and toposequences, mainly on a large scale (e.g. Bocquier, 1973 for Chad; Boulet, 1978 for Burkina Faso; Eschenbrenner and Grandin, 1970 for Ivory Cost and Burkina Faso and Turenne, 1977 for Guyana), but only a few studies consider the catenary variations on a smaller hillslope scale in terms of soil erosion (e.g. Gobin et al., 1999 for Nigeria; Mulders et al., 2001 and Stoop, 1987 for Burkina Faso). The advantage of the catenary concept is that variations in soil properties can be systematically included in any modeling approach in order to assess the magnitude of erosion and the optimization of land management practices on a hillslope scale (Kirkby et al., 1996). Recent modeling developments include such spatial concepts in their modeling approach to assess and distinguish the spatial distribution of erosion and deposition processes over the landscape more explicitly (De Alba et al., 2004; Märker, 2001; Mitas and Mitasova, 1998; Papiernik et al., 2009; Park et al., 2001; Schumacher et al., 1999; Wauw et al., 2008). Another advantage is that such delineation of hillslope or landscape units is based on a discrete approach in contrast to continuous, deterministic or stochastic methods, which require a large sampling number to derive conclusions (Burrough, 1993; McBratney, 2000; Park and Vlek, 2002).

On the methodological scale, such as observational and model scale, any soil erosion approach requires first of all a detailed understanding of the driving forces and underlying processes. Today, many general experimental studies exist especially for West Africa, which assess soil erosion on a filed scale by measuring the magnitude of soil loss on runoff plots (Casenave and Valentin, 1989; Le Bissonais et al., 1990; Diallo et al., 2004; Roose, 1977; Roose and Sarrailh, 1990). Great efforts have been made to gain insight into hydrological, physical and geochemical soil properties and to capture their influence on soil erodibility as well as their dependency on rainfall erosivity (Elwell and Stocking, 1976; Lafforgue, 1977; Lal, 1988; Roose et Le Long, 1976). This period of experimental field studies was useful to determine the mechanisms, functions and causes of soil erosion in natural systems and the interactions between individual parameters. Additionally, runoff studies have shown that erosion rates vary highly depending on specific soil characteristics and changes in slope gradients and vegetation cover, and elucidated that it is not possible to study all these variations in individual field experiments. Field investigations are time consuming and costly and cover only a small part of the landscape. Hence, they can neither account for more complex hydrological interactions nor for highly variable sediment flows and spatially distinct soil redistribution processes.

In order to overcome these spatial limitations of runoff-plots, empirical soil erosion models were developed relying on the results of experimental studies. The best known empirical erosion model is the Universal Soil Loss Equation (Wischmeier and Smith, 1978), which is still used – although in a modified form – for areas with limited data availability (Kinnell, 2010). Only a few erosion studies in West Africa are known that simulate soil erosion on a larger scale, such as hillslope, catchment or landscape scale (Veihe and Mati, 2001; Sterk et al., 2005). This might be due to the unsolved problems when dealing with different scales. In recent years, many physically based erosion models have been developed to quantify soil loss on a hillslope scale by considering the physical laws of energy and mass movement (e.g., EROSION 2/3D; EUROSEM; LISEM; MEDALUS; WEPP). The highly sophisticated models attempt to capture detailed small-scale processes as well as complex flow and routing algorithms by a large set of physical equations and mathematical functions. This leads often to an extremely complex model structure suffering from over-parameterization, exceedingly large data-sets and high computational requirements (Beven, 1989). The model's potential performance regarding accurate predictions is additionally limited by the high degree of uncertainty associated with input parameters, reflected in model outputs. Nevertheless, physical models are still far from simulating natural processes in all details and still neglect feedback mechanisms, non-linearities or emergent properties in their modeling approaches (Wainwright and Mulligan, 2004a). Yet, they have a good explanatory depth and are useful for gaining deeper knowledge about the process behavior of environmental systems.

However, the choice of the model will always depend on the guiding principle of parsimony, which says that "a parsimonious model is usually the one with the greatest explanation or prediction power and the least parameters or process complexity" (Mulligan and Wainwright, 2004, p. 8). Therefore, depending on the environmental conditions, the question of scale and the objectives for soil erosion assessment, simple empirical models may be considered a viable option and in specific conditions might be even more accurate, robust and reliable than highly complex models (Merritt, 2004; Perrin et al., 2001).

As validation of soil erosion models is fundamental, quantitative erosion and sedimentation measurements are needed. A more recent technique for determining the soil redistribution pattern is the measurement of the concentration of radionuclide ^{137}Cs in sediments (Mabit et al., 2008; Schumacher et al., 2005; van Oost, 2003; Walling and He, 2001; Zapata, 2002). This artificial isotope was produced by atmospheric bomb testing in the mid 1960s, distributed globally and deposited by rainfall on the earth surface (Walling and Quine, 1993). By comparing the ^{137}Cs concentration in undisturbed reference sites with those on sites where soil erosion or accumulation has occurred, medium-term erosion rates over the last 40-50 years can be calculated. The approach has already been applied in a few studies in West Africa (Chappell et al., 1999 in Niger; Faleh et al., 2005 in Morocco and Pennock, 2000 in Ghana) and might present an effective tool for measuring soil loss and for validating soil erosion models in Burkina Faso. Another option is to quantify sediments in reservoirs by bathymetric survey, which allows comparison of the initial with the more recent morphology of the reservoir bed. By knowing the sediment budget of the total catchment area, a better understanding of sediment flows and interaction on individual sub-scales is possible.

1.2 Research objectives

This research presents an integrated approach to quantify soil erosion and soil redistribution rates on different spatial scales in southwestern Burkina Faso by using scale-dependent pedological, morphological and hydrological methods, remote sensing techniques and soil erosion simulation models.

The main research objectives are:

- To assess the spatial dimension of soil erosion and soil redistribution rates at both hillslope and catchment scale in a semi-arid environment in southwestern Burkina Faso.
- To calculate siltation rates behind dams and sediment yield of catchments, and to identify the most strongly affected erosion and deposition zones (sediment source and sediment sink areas) within the landscape.
- To simulate the effect of feasible land management options on soil erosion at hillslope scale and to propose a soil erosion hazard map at the catchment scale.

1.3 Structure of the thesis

This first chapter states the problems related to soil erosion and reservoir sedimentation and emphasizes the relevance of soil erosion assessment at different scales. Additionally, the research objectives are outlined and the structure of the thesis is briefly described.

The second chapter reviews studies on soil erosion research in Burkina Faso, discusses the conceptual framework of different erosion models and refers to the considerable complexity and the issues of emergence and scale in environmental science.

The third chapter focuses on spatially distributed soil redistribution rates at the hillslope scale. Various research methods are presented to gain knowledge about systematic variations in soil properties (catena concept), to determine soil redistribution rates by ^{137}Cs measurements and to simulate soil erosion and the influence of land management strategies on soil loss by the Water Erosion Prediction Project (WEPP) model.

In the fourth chapter, soil erosion assessment is approached at a larger catchment scale. Investigations for calculating sediment yield in small dams by bathymetric survey and lake sediment analysis are evaluated, and the results of simulation scenarios by WaTEM/SEDEM are interpreted. Based on the latter, a remote sensing land-cover map is produced, and a digital elevation model is presented. Both serve as important input maps for the soil erosion and sediment delivery model. A soil erosion hazard map for the Ioba district is proposed to facilitate the identification of erosion and deposition hotspots.

Finally, in the fifth chapter conclusions are drawn and recommendations are given on how to deal with on-site and off-site effects of soil erosion and how to deploy the research results and the proposed soil erosion hazard map for future research needs.

2 STATE-OF-THE-ART

2.1 Soil erosion research in West Africa - Burkina Faso

A literature review of the historical background and development of soil erosion research in West Africa shows that research has undergone changes in scientific direction similar to those in many other academic disciplines. Both methodological approach and thematic focus have changed fundamentally within the last fifty to sixty years of erosion research.

Methodologically, soil research in West Africa moved from more descriptive, historical-genetic approaches with emphasis on mapping, classification and collection of basic data (1940-1965) to detailed process-based studies on experimental plots with the aim of measuring, explaining and modeling the functional relationships between system components (1965-1990). Technical improvements allowed widening the view from smaller to larger scales and from detailed process understanding to more generalizable methods. Since then, the focus has broadened in a multi-disciplinary pluralistic direction, and more applied approaches have developed mainly to be of local, regional or global relevance.

Thematically, the studies are closely related to these methodological concepts. While in the first phase, the need of fundamental knowledge about the soil per se was high, in the second phase more analytical experiments on geochemical, hydrological and physical soil characteristics took precedence. The basic scientific research was gradually extended to more globally and politically driven studies such as desertification, land degradation, sustainability, soil and water conservation. These in turn were then followed by concepts such as participation, regional planning, bottom-up approaches and, more recently, by climate change and carbon sequestration.

2.1.1 First perception of soil erosion

The problem of soil erosion was certainly recognized a long time ago by local farmers cultivating their soils in Burkina Faso. It is known, for example, that more than 300 years ago farmers in northern Burkina Faso already used “runoff-farming” techniques to irrigate their fields by canalizing rainfall water (Schmitt, 1987). Unfortunately further information is sparce, as the African culture is largely based on oral traditions and the first detailed documents reporting soil erosion were not found before the nineteenth century.

Early colonial studies, which were realized to gain basic knowledge about the structure, distribution and function of soils in West Africa (Aubert, 1951), recognized the important role of soil erosion (Aubert, 1948; Chevalier, 1948; Fournier, 1955). Especially the existence of laterite crusts ("cuirasse"), their characteristics and development was of interest to many researchers as early as the first half of the 20th century (Chetelat, 1938; Lacroix, 1913; Scaetta, 1938; Maignien, 1954). Aubert (1948) suggested to protect the soil against erosion and desiccation by covering it, thus preventing the irreversible formation of laterite crusts.

The influence of environmental factors on erosion such as precipitation, soil resistance, infiltration capacity, topography and micro-relief were already studied in 1956 by Fournier for soils on the Côte d'Ivoire, as well as the different forms and types of erosion such as sheet, rill, gully and tunnel erosion. He identified sheet erosion as one of the most important form of erosion. From measurements of soil loss on different agricultural plots he reported an annual soil loss rate between 6 and 27 t ha^{-1} depending on slope and vegetation cover. Considering that even low slope gradients of only 1.3%-2% caused high erosion rates, he concluded "catastrophic" soil loss rates on steeper slopes, which would have a severe impact on soil surface degradation (Fournier, 1956).

This interest in soil erosion led to the first studies on soil and water conservation with the aim to sustain the fertility of soils (Boyer, 1963; Jenny, 1965; Maignien, 1959; Mulard et Groene, 1961). Land capability maps were developed by Aubert and Fournier (1954) in which the land was classified according to land-use, value, capability and type of amelioration required for adequate use and conservation. Following a proposed reference scheme, land was assigned to specific land-capability classes depending on the soil quality, erosion susceptibility and topography. Soil conservation was introduced as a new element of soil assessment. Additionally, Fournier (1962) developed a "map of erosion danger in Africa south of the Sahara based on aggressiveness of climate and topography".

Further pedological investigations in the late 1960s and early 1970s focused mainly on soil genesis (Bocquier, 1971; Boulet et al., 1971; Kaloga, 1969), geomorphology (Avenard 1971; Boulet, 1970; Kaloga, 1970; Eschenbrenner and

Grandin, 1970) and soil fertility (Aubert, 1962; Boyer, 1970) and changed more and more from detailed soil profile investigations to a systematic classification of soil units and their distribution.

A French soil classification was developed (CPCS, 1967) and pedo-geomorphological maps for many West African countries were produced. Pedological maps on different spatial scales such as catchment scale (Kaloga, 1966; Maignien et. al, 1960) and landscape scale (Aubert, 1970; Boulet, 1968) were drawn for Burkina Faso, formerly Haute Volta.

At this time, large research networks were established (e.g. by ORSTOM[1], GERES[2], CTFT[3]) and many measuring stations were installed around West Africa to collect hydro-meteorological data (Dubreuil, 1971; ORSTOM, 1956) and soil loss data from runoff-plots (Birot et al., 1968; Dabin, 1959; Fournier, 1954; Roose, 1965).

2.1.2 Process-based studies and experimental plots

Soil erosion research benefited highly from the large measuring networks and many experiments on runoff-plots for quantifying the effects of soil properties, slope gradients, vegetation cover and agricultural land-use on soil loss. The first runoff-plot was set up in 1954 at Sefa, Sénégal, and functioned from 1954-1969. Others followed in Burkina Faso (Niangoloko: 1956-1957; Gampela: 1968-1979; Saria: 1971-1974), Côte d'Ivoire (Adiopodoume: 1956-1976; Bouake: 1960; Korhogo: 1972-1975) and Guinea (Kindia: 1956-1958, Seredou: 1955-1956) (Roederer, 1975). Later, the network was extended and experimental stations in Niger, Nigeria, Benin and Ghana were implemented by other research organizations or individual scientists (Collinet et Valentin, 1980; Lal, 1976; Roose, 1977). About 560 experimental plots are known for West Africa (Roose, 1994).

Especially, the erosive force of rainfall as the main agent causing soil erosion was analyzed thoroughly using rainfall simulators (Lafforgue, 1977; Roose, 1973; Roose et Le Long, 1976). The aggressiveness of rainfall depends on its magnitude, intensity, kinetic energy and duration. The relationship between these parameters was

1 Office de la Recherche Scientifique et Technique Outre-Mer (today IRD: Institut de Recherche pour le Développement)
2 Groupement d'Etudes sur la Restauration des Sol
3 Centre Technique Forestier Tropical (today CIRAD: Centre de Coopération Internationale en Recherche Agronomique pour le Développement)

evaluated by many scientists for African environments (Brunet-Moret, 1967; Charreau, 1970; Elwell and Stocking, 1976; Lal, 1981; Roose, 1980) and finally an index was accepted expressing rainfall erosivity as a function of kinetic energy and maximum intensity of a 30-min rainfall event. This simplified index - based on the proposed equation of Wischmeier (1959) - was slightly modified for conditions in West Africa by adding coefficients derived from measured data from meteorological stations (Charreau, 1970; Delwaulle, 1973; Gallabert and Millogo, 1973):

$$R = 0{,}01572 H \times I_{30} - 1.179 \tag{2.1}$$

where R = rainfall erosivity, H = magnitude of rainfall and I_{30}= maximum intensity of a 30-min rainfall event (Roose, 1977, p. 31)

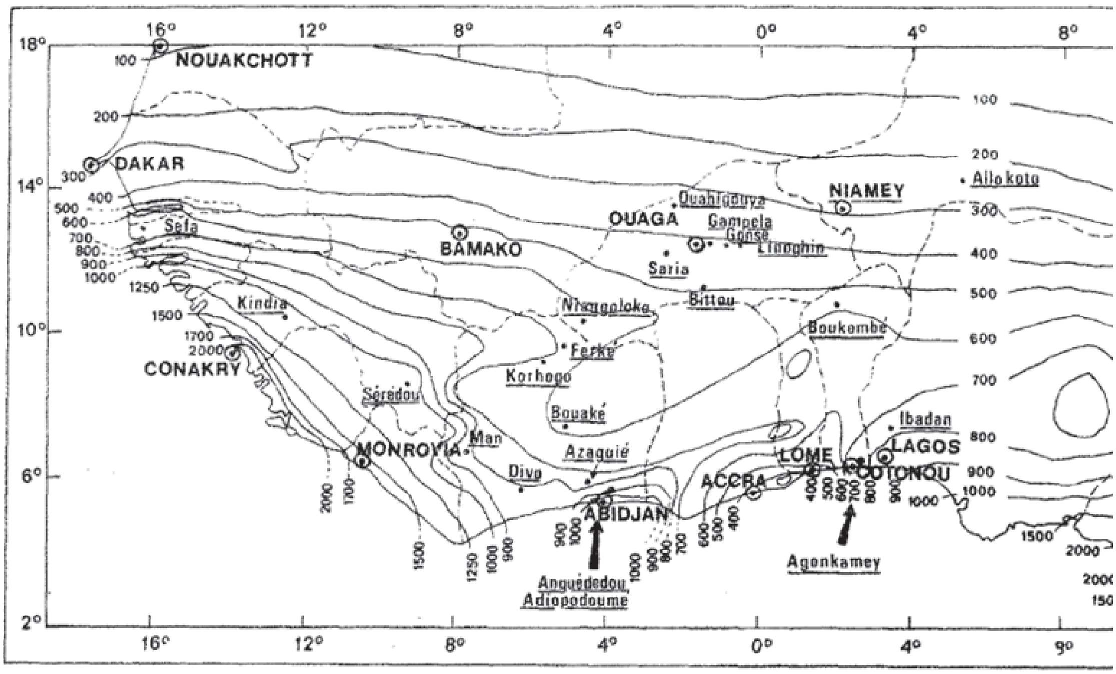
In order to transfer this index to other regions where no meteorological data were available, Roose (1977) proposed a rainfall erosivity map for West Africa (Figure 2.1). This map shows the distribution of erosivity based on the index of Wischmeier (1959), mean annual precipitation and the distance to either coast or mountains. Rainfall erosivity (R) is almost parallel to the isohyets and increases from 100-200 in Sahelian regions to 400-600 in savannah environments (Ouagadougou, R = 430 and 830 mm rainfall) and to more than 1000 in the humid tropics (Abidjan, R = 1.200 and 2.100 mm rainfall). Initially drawn as a sketch for further discussion, the erosivity map was critically reviewed, modified and adjusted (e.g. El-Swaifi and Dangler, 1982) and finally used as reference in many soil erosion studies.

Another concern of experimental studies was to gain knowledge about the erodibility of soils. The resistance of a soil against erosion depends mainly on internal soil characteristics, vegetation cover and land management practices. Although soil resistance is generally higher in tropical regions than in temperate zones (Nill, 1993; Martin, 1988; El-Swaifi and Dangler, 1982), it is closely linked with rainfall intensities, which explains the high erosion rates in African environments.

Early erodibility studies in West Africa were influenced first of all by the dispersion test of Middleton (1930) and the diagram of Hjulström (1935), who explained soil detachment, transport and deposition as a function of water speed and soil particle size.

Esquisse de la répartition de l'indice d'agressivité climatique annuel moyen (R_{USA} de Wischmeier) en Afrique de l'Ouest et du Centre

Situation des parcelles d'érosion (lieux soulignés)



D'après les données pluviométriques rassemblées par le Service Hydrologique de l'ORSTOM et arrêtées en 1975

0 400 300
200 600 1000 Km

Dresée par Roose (E.J.)

Figure 2.1 Rainfall erosivity map for West Africa (Roose, 1977, p. 33, extract)

Later studies considered the aggregate and structural stability tests of Ellison (1944 and 1954) and Hénin (Hénin and Monnier, 1956). Laboratory experiments were performed to analyze aggregate stability and particle size detachment as a function of raindrop energy and cohesive soil characteristics, the so-called "splash effect" (Fournier, 1948; Hudson, 1963; Quantin et Combeau, 1962).

In a further development, process-based studies emerged which tried to capture physical and geochemical soil parameters from laboratory experiments and microscopy in order to explain micro-morphology and mineralization mechanisms in soils (Grandin, 1976; Lelong et al. 1976; Leprun, 1977; Millot et al., 1976). Later, field studies using rainfall simulators focused mainly on the influence of organic matter, soil texture, $CaCO_3$, hydro-oxides and infiltration capacity on surface runoff and soil loss (Albergel et al., 1986; Casenave and Valentin, 1989; Collinet, 1988; Collinet and Valentin, 1979; Dumas, 1965; Roose, 1965; Roose and Sarrailh, 1990).

Due to the wide range of influencing factors, no strict correlation exists between soil type and soil erodibility, but certain relationships can be derived. For example, in the French classification scheme (CPCS, 1967) sols ferrugineux tropicaux (which would correspond to lixisols ferric in the FAO-classification scheme) are relatively fragile and measured erodibility values vary between 0.2-0.3, whereas sols ferralitique (FAO: lixisols rhodic) are less fragile and have values of 0.01-0.2. Especially sols gravillonnaires (FAO: leptosols skeletic) and lithosols (FAO: plinthosols) as well as vertisols calciques (FAO: vertisols calcic) are more resistant to erosion and have values of 0.01-0.04 and 0.001-0.01, respectively (Roose, 1976 and 2004).

Whereas rainfall erosivity is an unchangeable natural factor, and hence is an independent variable, the erodibility of a soil can be partially influenced by land management methods. Suggestions for improving the resistance of soils are either to cover the soil permanently by vegetation or at least residues, to increase soil organic matter content by mulching, or to protect the soil by gravel and stone lines (Lal, 2001; Roose and Sarrailh, 1990; Dumas, 1965).

However, the impact of rainfall erosivity and soil erodibility on erosion has not yet been satisfactorily solved and still attracts the attention of many scientists as recently shown by the increase in experimental studies using rainfall simulators (Le Bissonnais et al., 1990; Diallo et al., 2004; Lal, 1988; Planchon, 1991; Rodier, 1992; Valentin and Janeau, 1990). The period of intensive field studies so far was useful to determine the mechanisms, functions and causes of natural systems and the interactions between individual parameters, e.g., rainfall intensities related to infiltration capacity, or raindrop energy related to particle size detachment. Additionally, runoff studies have shown that erosion rates vary highly depending on specific soil characteristics, small changes in slope gradients and vegetation cover, and that it is not possible to study all these variations by individual field experiments. Furthermore, the set up and maintenance of experimental plots is very time consuming and costly.

New approaches evolved in order to capture the functional relationship between parameters with physical laws and to formulate the interactions between the main driving forces. The advances in computer science and thus the development of soil erosion models was a decisive step forward.

2.1.3 Soil erosion modeling

The Universal Soil Loss Equation (USLE), developed by Wischmeier and Smith (1960; 1965, 1978) for soils in the United States, was already cited in 1969 by Fournier. He emphasized the role of climate in influencing soil erosion in West Africa, while other factors, such as soil, topography and vegetation cover were considered of secondary importance.

Roose (1976) was one of the first scientists to check the applicability of the USLE in West Africa by evaluating soil loss and runoff data from about 50 experimental plots located in five countries (Figure 2.1). The experiments covered a large variety of environmental conditions, including a south-north climate gradient ranging from 2100 mm average annual rainfall in the South of the Côte d'Ivoire to less than 500 mm in the Sahelian region of Niger. Slope gradients varied from 1% to 65%, soil types were mainly ferralic and ferric tropical soils, and vegetation cover ranged from bare soil and a variety of agricultural crops to typical savannah vegetation and dense forest (Table 2.1). Results showed that annual soil erosion cannot solely be explained by the amount of rainfall and its intensity (R-factor). Slope variations (SL-factor), soil erodibility (K-factor), vegetation cover (C-factor) and support practices (P-factor) have also to be considered. For example, slope gradients increasing from 7% to 20% in Adiopodoume (Côte d'Ivoire) led to three to four times higher erosion rates under natural vegetation and bare soil, respectively. Erosion rates increased by more than four thousand times from 0.03 t ha-1 under natural vegetation to 138 t ha-1 for bare soil with slope gradients of 7% and from 0.1 t ha-1 to 570 t ha-1 with slope gradients of 20%. Although these extreme values present the maximum soil loss measured, other experimental studies also indicate a high influence of vegetation cover. Generally, soil loss can be reduced by 20-60% if bare soil is covered by field crops (Roose, 2004).

Therefore, increasing attention was paid to vegetation cover as an important factor in land management practices. Vegetation cover also plays an important role regarding soil erodibility and thus resistance to erosion as mentioned earlier (Roose and Sarrailh, 1990). The following values are recommended as C-factor values for the USLE in West Africa: 1 for bare soil; 0.1-0.9 for agricultural crops such as corn, sorghum, millet (0.4-0.9), cotton, tobacco (0.5-0.7) and rice (0.1 – 0.2); 0.1 for over-grazed savannah or prairie, and 0.001 for forest (Roose, 1976 and 2004).

Table 2.1 Erosion (t ha^{-1} yr^{-1}) from experimental stations in West Africa

Location	Rainfall (mm)	Soil type	Slope (%)	Erosion (t ha^{-1} yr^{-1}) Natural environment	Bare soil	Cropped soil
Adiopodoumé (Côte d'Ivoire)	2100	leached ferralic	4.5	-	60	-
			7	0.03	138	0.1 to 90
			20	0.1	570	-
			65	0.2 to 1	-	-
Anguédédou (Côte d'Ivoire)	2000	leached ferralic on tertiary sandy clay	29	-	-	0.6 to 0.3
Anzaguié (Côte d'Ivoire)	1800	ferralic on schist	14	0.5 to 0.7	-	0.9 to 4.6
Divo (Côte d'Ivoire)	1750	ferralic on granite	9	0.5	-	-
Bouaké (Côte d'Ivoire)	1200	eroded ferralic on granite	4	0.2 [(a)]	18 to 30	0.1 to 26
				0.01 [(b)]	-	-
Korhogo (Côte d'Ivoire)	1400	ferralic on granite	4	0.1 to 0.2 [(a)]	3 to 9	-
Ouagadougou (Burkina Faso)	850	leached ferric on granite	0.5	0.15 [(a)]	10 to 20	0.6 to 8
				0.01 [(b)]	-	-
Sefa (Senegal)	1300	leached ferric	1-2	0.02 to 0.5 [(a)]	30 to 55 [(a)]	
				0.02 to 0.2 [(b)]		2 to 20 [(b)]
Cotonou (Benin)	1300	leached ferralic on tertiary sandy clay	4	0.3 to 1.2	17 to 27.5	10 to 85
Boukombe (Benin)	1100	leached gravel-filled ferric on schist	3.7	-	-	0.2 to 1.6
Allokoto (Niger)	500	calcic vertisol	3	-	-	0.1 to 18.5

[(a)] burned *[(b)] not burned* *(modified from Roose, 1976, p. 68)*

Land management practices like mulching, ridging along contours, buffer grass strips or stone lines have often been found to improve soil structure and to reduce runoff and soil loss as shown in many studies (Combeau, 1977; Kiepe et al., 2001; Lafforgue and Naah, 1976; Lal, 2000; Mietton, 1986; Roose, 1994 and 1998). In northern Burkina Faso, Lamachère and Serpantie (1990) found that stone lines/bunds (*cordon pierreux*) improved the hydraulic balance of soils, mainly by increasing the infiltration capacity by an average of 15% on sandy to sandy-clay soils. Additionally, Zougmoré et al. (2004a) noted that runoff was reduced by 45% when stone lines were constructed and by 53% when grass strips (*bandes d'arrêt enherbées*) were used on leached ferric lixisols. Roose (1976) concluded that artificial mulch was most effective in reducing soil loss by 40%-95% and that straw mulch (*mulch de paille*) of only a few centimeters thickness had the same effect as a secondary forest of 30 m height in absorbing kinetic rainfall energy. Therefore, a relatively high support practice value (P-factor) of 0.2-0.5 is proposed for artificial mulch (but only 0.01 for straw mulch), whereas buffer grass strips are assigned a value of 0.1-0.3, stone lines and earth ridges 0.1 and ridging along contour 0.1-0.2 (Roose, 1994).

These values indicate that biological management practices are generally more effective in reducing soil erosion than mechanical soil and water conservation techniques, which are often recommended without considering the high implementation and maintenance costs. Another benefit of biological practices is their contribution to soil fertility due to increased organic matter, enhanced mineralization and increased biological activity in an aerated humus layer. However, the bottleneck of biological practices such as mulching is the difficulty to produce sufficient quantities of vegetative residue for covering the fields adequately. This is often not possible without biomass transfers. An alternative would be the use of expensive artificial mulch.

Continuous efforts to improve individual factor values for the USLE by intensive field and laboratory experiments and progressive attempts to update its specific equations and nomographs have made the USLE the most frequently used soil erosion model worldwide. A major advantage of the USLE is its simplicity and the availability of a large empirically based data set, which is why the model is applied until today by many scientists to a wide range of environmental conditions.

In West Africa, for example, the applicability of the USLE was evaluated by Veihe and Mati (2001), who concluded that the model can be used for savannah environments if some degree of calibration and validation is provided. Nevertheless, they detected some shortcomings in factor values for vegetation cover and land management practices (C and P factors) and suggested developing more appropriate indices to consider intercropping, hand-tilled crops, agroforestry and rangelands as well as local soil conservation techniques, such as stone lines.

In more recent studies, the USLE has been used for erosion risk assessment on larger scales using geographical information systems (GIS) and remote sensing techniques (Anys et al., 1994; El Garuouani et al., 2003; Vrieling, 2006). Scientists have taken advantage of the simple factorial model structure, which allows deriving each factor independently and integrating them as individual layers into GIS. Land-cover information can thus be derived from satellite images or aerial photographs, which facilitates the coverage of larger areas. Bonn (1998), for example, used GIS and remote sensing techniques for soil erosion assessment in Morocco and validated soil loss simulated by the USLE, first with soil loss measurements from accumulated sediments in reservoirs, then with soil loss estimations by the ^{137}Cs technique, and finally with sediment tracer studies using magnetic soil characteristics as indicators for soil loss. A comparison of these different quantification methods showed that average annual soil loss rates for the catchment were in the same order of magnitude, namely 39 t ha^{-1} for the USLE, 43 t ha^{-1} for accumulated lake sediments and 48 t ha^{-1} for ^{137}Cs measurements and also corresponded well with sediment source areas identified by magnetic soil properties.

Nevertheless, considerable errors are related to the model approaches both due to the simple multiplicative equation of the USLE, where each factor is independently accounted for as an isolated part of the landscape, and due to uncertainties for individual factor values (Bonn, 1998; Burrough, 1986;, Kinnell, 2010). Especially for land-cover values derived from remote sensing techniques, where the resolution of the satellite images (e.g. Landsat with a pixel size of 30x30m) is often equal or even below farmers field plot size, the accuracy of factor values is low and cannot reflect the fragmented landscape pattern typical for small-scale farmers in West Africa (Mati and Veihe, 2001). Furthermore, the model neither considers dynamic interaction between factors nor feedback mechanisms. Also, scaling effects are not accounted for, which might increase potential errors (Kirkby, 1998; Gray, 1999). Therefore, soil erosion results obtained by the USLE should be interpreted with care and may be better used as relative estimates to identify erosion hazard zones and sediment source areas rather than as absolute values.

Further modeling attempts were made using other empirically, but also physically-based soil erosion models. For example, Igwe et al. (1999) applied the Soil-Loss Estimation Model for Southern Africa (SLEMSA, developed by Elwell, 1978; Elwell and Stocking 1982) for erosion hazard mapping in Nigeria and assessed its overall predictive ability as good. However, compared with USLE, he found that the model was less able to reflect actual soil erosion patterns on a large scale. In addition, Visser et al. (2004) used the physically-based European Soil Erosion model (EUROSEM, developed by Morgan et al., 1992) for runoff and soil loss prediction on a field scale in northern Burkina Faso. Although the model does not consider crust development and its related effects on infiltration and detachment, she found a good agreement between simulated and measured runoff and sediment discharge results from experimental plots and concluded that a rewritten version of EUROSEM for Sahelian environments is suitable to simulate infiltration, runoff routing, sediment transport, erosion and deposition processes for individual storm events (Visser et al., 2004; Sterk et al., 2005).

So far, neither empirical models, which rely on measured values from field experiments, nor physically-based models, which require a high number of input parameters, could satisfactorily solve the problem of quantifying soil loss in West Africa. As almost all soil erosion models were originally developed for different environments and later adopted to other regions, detailed calibration and validation procedures are indispensable. Uncertainties remain and recent trends in soil erosion studies show that more applied approaches to assess and prevent soil erosion are emerging with a focus on determining the economic costs related to soil erosion.

2.1.4 New “fashions”, policy interventions and pluralistic views

Sperber (1990) describes the directions and development in science by changes in methods, goals or philosophies within a discipline with the term “fashion”. The pathway of a discipline is directly related to the emergence of new paradigms, opinions or current leader – so-called “dude”, who sets the new fashion (Sherman, 1996). The concept of *fashion change* can also be applied to developments and trends in soil erosion research.

As the above review has already shown, research has moved from a positivistic view, based on empirical data and inductive approaches, to a post-positivistic view which is characterized by process-based studies and deductive, theoretical concepts trying to quantify, simulate and predict soil erosion by models (Figure 2.2). The conventional and empirical direction in ecological research can also be interpreted as “techno-centric with elements of positivism and reductionism”, which was replaced by detailed process understanding or so-called *system-thinking* approaches defined as "eco-centric with elements of positivism and holism" (Stroosnijder and Rheenen, 2001, p. 389).

Since then, research has shifted more and more to applied science reflecting a "holistic-centric approach (…) with elements of holism and constructivism" (Stroosnijder and Rheenen, 2001, p. 389). Studies based on this new approach were often characterized by problem-oriented, participatory objectives or soil conservation issues focusing on local, regional, but also global concerns, such as land degradation, desertification (Gray, 1999; Koning and Smaling, 2005; Mortimore and Turner, 2005; Warren, 2001) or more recently carbon sequestration (Perez et al., 2007; Roncoli et al., 2007; Roose et al., 2006; Van Oost, 2007; Yadav et al., 2009). Simultaneously, soil erosion research has been influenced by concepts of realism, and studies had to be justified with regard to their effectiveness, practicality and feasibility in the achievement of ecological and socio-economic goals. In this context, many multi-disciplinary studies emerged dealing with environmental sustainability, rehabilitation of natural resources, soil fertility, social and cultural responsibility, economic cost-benefit analysis, food security, livelihoods, and the consequences of globalization and climate change (Fairhead and Scoones, 2005; Heerink, 2005; Sivakumar, 2007; Smaling and Dixon, 2006; Warren, 2005).

The resulting pluralistic palette of scientific studies is too diverse to be discussed in detail. Therefore, only a few important developments in soil erosion research are mentioned here. The direction in science was first influenced by the goals and strategies of international conferences and political interventions that set priorities for research projects. In 1977, the United Nations Conference on Desertification (UNCOD) in Nairobi addressed for the first time the problem of desertification and could be seen as the starting point for many soil conservation programs and action plans combating land degradation worldwide (UNCOD, 1978; FAO and UNEP, 1984; UNDP, 1999). Special emphasis was placed on Sahelian regions, which were severely affected by droughts between 1968 and 1973 (Jean, 1975; Sircoulon, 1976) and in the early 1980s (Valentin, 1994).

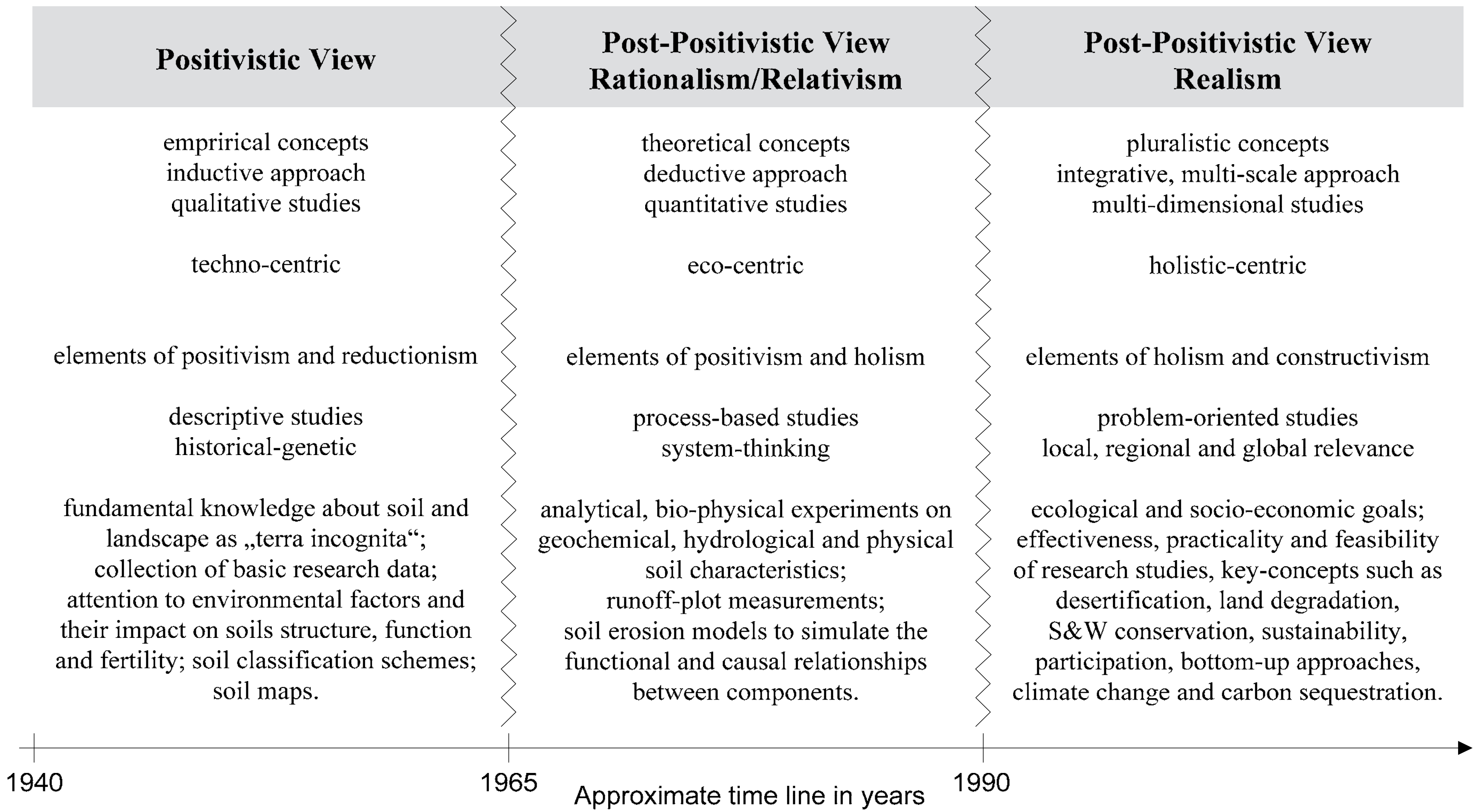


Figure 2.2 Scientific directions influencing soil erosion research (based on Brown, 1996; Rhoads and Thorn, 1996a; Stroosnijder and Rheenen, 2001)

In Burkina Faso, many research programs were realized to support natural resource management and soil and water conservation techniques (e.g. PNLD[4]; CILSS[5]).

At the United Nations Conference on Environment and Development (UNCED) in 1992 in Rio de Janeiro, it was realized that despite continuous efforts and noticeable local improvements to counteract land degradation, global desertification problems were increasing. Therefore, an international action program was agreed on to implement the goals of the Agenda 21 to combat desertification and droughts (BMZ, 1999). The claim for new integrative strategies emphasizing sustainable development ("Brundtland-Report": WCED, 1987), participation (Chambers, 1994) and decentralization were key issues at the convention and encouraged policy makers and scientists to shift rural communities and local farmers into the center of soil conservation projects and land-use planning. At the Panafrican Conference, held 1997 in Ouagadougou by the UN Convention to Combat Desertification (UNCCD), soil erosion was identified as a major cause for physical degradation of land. Furthermore, action was claimed to counteract hardpan formation, which limits infiltration and increases runoff.

In West Africa and specifically in Burkina Faso, these political interventions can be seen as an important motivation to widen the focus of soil erosion research and to integrate new concepts, such as sustainable land management (Veihe, 2000; Warren, 2002), participatory decision making (de Jager, 2005; Hessel et al., 2009; Mazzucato and Niemeijer, 2000; Nederlof et al., 2001) and community-based rural planning (Battergury, 1994; Sedogo and Groten, 2000), into many qualitative studies.

One well established and widely spread community-based approach for rural development in West Africa is the *Programme National de Gestion de Terroir* (PNGT, 1993). The term *terroir* is defined here both in its physical-ecological context as an area of land to be cultivated and managed, and in its social significance as a space for living, social-political interactions and economic interests (Sedogo, 2002; Teyssier, 1995). As the responsibility to use, manage and maintain natural resources is in the hands of farmers, the local community should not be seen any longer only as a target group, but as empowered actors involved in decision-making processes. Such empowerment of the local population through participation from the beginning of the project may lead to

4 Programme National de Lutte contre la Désertification
5 Comité permanent inter-etats de lutte contre la sécheresse dans le Sahel

new incentives for the formulation of needs, planning procedures and implementation strategies and may possibly produce more ecologically and socially sustainable results. This change from formerly strictly centralized and highly technocratic top-down programs to decentralized participatory rural planning with farmers who are agents and beneficiaries at the same time is reflected in many development projects (e.g., CES/AGF[6], PATECORE[7]; PSB[8] see also CCD, 2004; Reij and Thiombiano, 2003; GTZ, 2005).

Additionally, local soil and water management techniques are being reconsidered. Both, traditional and modern techniques play an important role in the advance of soil erosion research in West Africa. One well known and promising example of a traditional and recently re-animated land management practice is the zaï technique (or the modified zipellé technique), where organic material is mixed into small planting/water harvesting holes in order to restore soil fertility and to improve the nutrient supply in degraded or crusted soils (Fatondji; 2002; Kiepe et al., 2001; Roose et al., 1999; Zougmoré, 2004a). Other scientists compare farmers' perception of soil erosion and land degradation with scientific approaches (Gray and Morant, 2003; Hessel et al., 2009; Niemeijer and Mazzucato, 2003) or look at local knowledge in assessing soil fertility change (Fairhead and Scoones, 2005; Warren et al., 2003) to gain a better understanding of how local farmers use and manage their soils.

This increasing interest in local knowledge systems is further reflected by the recently-introduced hybrid discipline ethnopedology, which combines multi-perspective views and methodologies from social and natural science (Barrera-Bassols and Zinck, 2003). It includes the formalization of local knowledge into classification schemes and its comparison with scientific soil taxonomies, the analysis of local land evaluation systems from a socio-cultural and from a bio-physical perspective, and the assessment of land management practices, both modern and traditional. Three main domains are identified representing the theoretical framework of the concept, namely the symbolic (*kosmos*), cognitive (*corpus*) and the management (*praxis*) dimension - so-called K-C-P complex or ethno-ecological model. The term *kosmos* describes the perceptions of the local environment as influenced by belief systems, symbolic meanings and socio-

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[6] Programme spécial Conservation des Eaux et des Sols et Agroforesterie
[7] Project Aménagement des Terroirs et Conservation des Ressources
[8] Programme Sahel Burkinabé

cultural or sacred values. The *corpus* components includes a more cognitive understanding of natural resources expressed by indigenous knowledge systems, soil taxonomies and land classification schemes. The *praxis* component addresses land-use and land management strategies as well as soil and water conservation techniques (Barrera-Bassols and Zinck, 2003; Barrera-Bassols et al., 2006).

Beside this extreme ethno-anthropological view, there is the more profane, secular view of land degradation reflected in many economic studies, which consider incentives and constraints of farmers to invest in natural resources from an economic monetary perspective (Knowler, 2004; Koning and Smaling, 2005; Warren, 2001). Often, farmers' willingness to invest in sustainable soil and land management depends on the financial returns and agro-ecological benefits of soil conservation measures. From a long-term, global perspective, Koning and Smaling (2005) concluded that public investments and a reversal in price policies are needed to support, on the one hand, farmers' decisions toward sustainable agriculture and to avoid, on the other hand, agricultural intensification at the expense of natural resources.

Warren (2002) summarizes these dynamical shifts in viewing soil erosion in his statement "Land degradation is contextual". He holds the view that "land degradation cannot be judged independently of its spatial, temporal, economic, environmental and cultural context" (Warren, 2002, p. 449). Therefore, it is preferable that many multi-dimensional and inter-disciplinary studies contribute to soil erosion research in order to ensure integrative holistic approaches including different spatial and temporal scales and hierarchical levels. The call for more applied research and increasing political awareness was formulated by the "Millennium Development Goals" (UN Millennium Project, 2005) and resulted in the declaration of the year 2006 as the "International Year of Deserts and Desertification".

Nevertheless, besides these changing fashions and new paradigms, which directly effect science, basic research needs should not be neglected. Fundamental basic approaches should be seen as a complement for applied research rather than as a competing or outmoded scientific approach, because "all scientific knowledge has practical value, albeit perhaps in a highly indirect and unforeseeable manner" (Rhoads and Thorn, 1996b, p. 132).

2.2 Scientific nature of soil erosion models

No model per se can reflect reality in all its details. Any model suffers from internal limitations due to its designed structure and remains imperfect when representing real processes. But, as models can help us understand, analyze and deepen our knowledge about environmental systems and their interactions, the question remaining is: “Which model to use”?

2.2.1 Models as representations of environmental systems

Models provide a tool to analyze and predict the behavior of natural systems. Any system is characterized by a specific flow of mass and/or energy, which enters the system as an input, is transmitted through the system by interaction with internal and external variables, and leaves the system as an output. Variables within the system can be presented on the one hand as objects, defined as parts or components of a system which are either physical (e.g., soil or plants) or abstract (e.g., equations and processes) and on the other hand as attributes, defined as physical properties of the objects (Thorn, 1988). Changes in inputs usually produce changes in outputs. Most natural systems can be classified as open systems, which permit a movement of energy and mass across the system’s borders. In contrast to open systems, isolated systems have boundaries that prevent the import and export of either mass or energy, whereas closed systems permit only energy but no mass movement throughout the system (Thorn, 1988).

The variables of a system, which control the flow of mass and energy, can be either independent (causal) or dependent (responding to causal variables) (Summerfield, 1997). Variables are quantities that affect the system being studied. Variables can be internal or external to the system. Internal variables are quantities or flows as a result of defined processes within the system, e.g., soil water content, whereas external variables are not system immanent and influence the system only partly by controlling its functioning, e.g., rainfall (Young, 1994). The term parameter is used for fixed values, which are unchanged, either for all cases or for a defined set of conditions. In this study, the term parameter describes a fixed dataset of required inputs, which refers consequently to external or internal variables, which are changed within the modeling procedure as a result of internal mathematical functions.

Systems can be classified by the relationship of variables within the system. In geomorphological approaches, three kinds of systems can be identified with increasing level of complexity: *Morphological systems*, which describe morphological variables by size, shape, etc., and consider the statistical relationship between morphological properties and landform elements; *cascading systems*, which additionally consider the flows of energy and mass through the system; and *process-response systems*, which analyze the interaction between variables and processes of the system and consider the output of one part of the system as an input to another part of the system. These process-response interactions are called negative or positive feedback, depending on its decreasing or increasing effect on the system (Thorn, 1988).

Whereas a system can be defined as an abstract concept of "any set of interrelated or interconnected elements, (…) assumed to exist in the real world" (Strahler, 1980, p. 1), a model is a simplified representation of some part of reality (Grayson and Blöschl, 2000). This implies that even a highly complex and sophisticated model will always be partially inadequate and consequently incorrect to some degree. Nevertheless, as a simplified representation of reality, a model can be used to estimate changes of energy and mass entering and leaving the system, which might enable the user to understand relationships and interactions between system variables and to determine the behavior of a system under changed conditions. Every model focuses only on a fragmentary part of a natural phenomenon and thus has to set priorities depending on its objectives in order to analyze, simplify and computerize processes.

A model should obey the principles of parsimony ("a model should not be any more complex than it needs to be and should include only the smallest number of parameters whose values must be obtained from data" Hillel, 1986, p. 42), modesty ("a model should not pretend to do too much" Hillel, 1986, p. 42), accuracy ("we need not have our model depict a phenomenon much more accurately than our ability to measure it" Hillel, 1986, p. 42) and testability ("a model must be testable" by considering "what are the limits of its validity" Hillel, 1986, p. 42). Especially the principle of parsimony should be reconsidered in an era where technical skills allow highly complex and often over-parameterized model structures. During the conceptualization of the model, the scientist determines which part of reality should be represented by the model, which key processes should be analyzed in detail, which dominant variables control the response, and which processes could be simplified or neglected. The temporal and spatial scale of a model depends on its conceptual framework and the theory behind it (Ford, 2000).

In this context the model can be described as "a mathematical expression of this hypothesis and is in a form that can be tested" (Grayson and Blöschl, 2000, p. 52) or, in other words, modeling can be seen as the "art of constructing mathematical models and the study of their properties for comparison with the properties of the systems they represent" (Young, 1994, p. 423).

2.2.2 Basic concepts of soil erosion models

A wide range of erosion models has been developed to meet the requirements of specific objectives, intentions and applications in regard to various environmental conditions and different spatial and temporal scales (Merritt et al., 2003; Boardman, 2006). Figure 2.3 illustrates one possibility of categorizing models in terms of their conceptual framework, input specification, and temporal and spatial representation (see also Ford, 2000; Grayson and Blöschl, 2000; Hardisty et al., 1998; Hillel, 1977).

Concepts of soil erosion models are based either on empirical studies (e.g., USLE) or on physically-based equations (e.g., WEPP). Although a large number of measurements were performed to determine input factor values for distinct environmental settings, empirically based models remain time and space dependent. They do not consider functional or causal relationships between system components and are mainly based on statistical methods (e.g., regression analysis). More recent soil erosion models use physically-based or process-based algorithms, which quantify soil loss as a result of energy flows and mass movement depending on physical laws. The term process-based refers to the ability of models to reflect dynamic processes, whereas the term physically-based refers to internal equations for calculating processes mathematically based on physical laws. Both types of models are not able to reflect non-linear responses and complex feedback mechanisms of a system and frequently rely on empirical equations or at least coefficients to simulate erosion processes (Morgan, 1995).

Furthermore, erosion models can be divided by type into deterministic or stochastic models. In the case of deterministic models, a single set of input values is used to generate a single set of outputs (unique solutions).

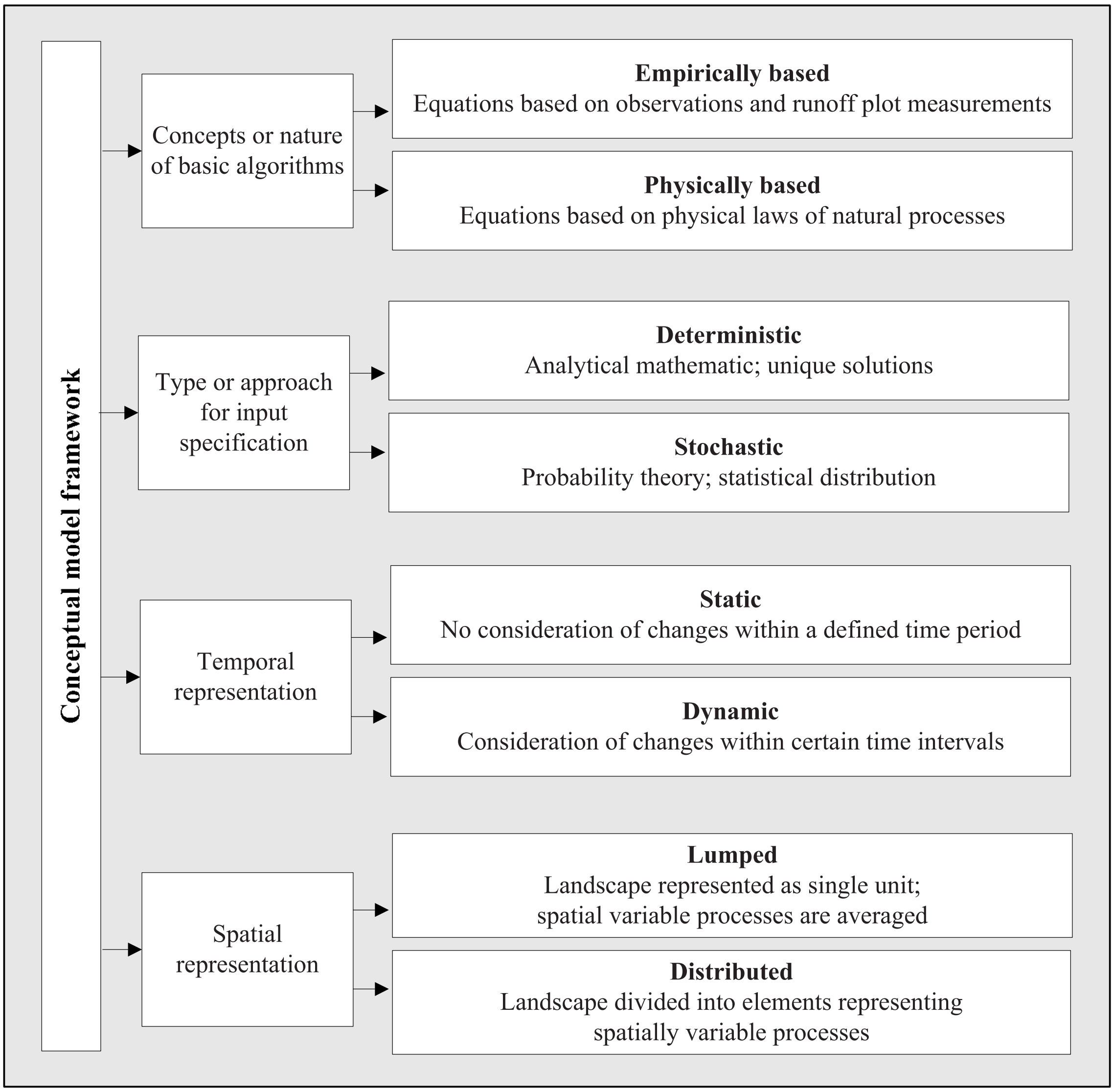


Figure 2.3 Conceptual framework of models

This way an unambiguous cause-effect relationship between components is identified. Stochastic models deal with a certain probability of occurrence, which means that input values are represented by statistical distributions rather than by a single value. As a given set of inputs can produce a range of output responses, the stochastic approach is often preferred in terms of uncertainties in spatial and temporal data or for sensitivity analysis (Grayson and Blöschl, 2000). Concerning the temporal resolution, static models can be differentiated from dynamic models. Static models describe processes within a system over one fixed time interval, while dynamic models represent changes over time within the system. For the later, time is considered as a continuous and independent variable.

In their spatial representation, lumped models can be distinguished from distributed models. Lumped models are not spatially explicit, which implies that the complete landscape, e.g., hillslopes, is considered a single unit, and processes are averaged over space. Distributed models offer an option to integrate spatial variability by dividing the landscape into individual elements representing specific processes within these elements, e.g., slope positions along a hillslope catena. By increasing the number of landscape elements over space, a higher number of individual processes can be reflected and a higher spatial representation is reached (Grayson and Blöschl, 2000).

Conceptual models (e.g. AGNPS) play an intermediate role between empirically and physically-based models. They reflect the dynamic behaviour of a system by considering internal storages, transfer mechanisms of sediment, runoff generation and flow paths. They include a general description of catchment processes (e.g., internal storages, flow paths) without including the specific details of process interactions (Merrit, 2003).

2.2.3 Empirically and physically-based models: Advantages and limitations

The selection of an appropriate soil erosion model depends on the scope of the study and the ability of the model to serve as a tool to solve the research questions. It involves considerations about the conceptual framework, major components and advantages and limitations of the model, but also about the adequate temporal and spatial scale on which soil erosion is simulated (Boardman and Favis-Mortlock, 1998; Boardman, 2006; Bork and Schröder, 1996; Jetten, 1999; Kinnell, 2010; Merritt, 2003; Schmidt, 2000; Wainwright and Mulligan, 2004b).

The first soil erosion modeling approach, the USLE developed by Wischmeier and Smith (1978), is a simple multiplicative model based on the mathematical algorithm:

$$A = R \times K \times L \times S \times C \times P \tag{2.2}$$

where A = average soil loss per unit area; R = rainfall-runoff (erosivity) factor; K = soil erodibility factor; L = slope length factor; S = slope steepness factor; C = cover management factor and P = support practice factor (e.g., contouring, terracing) (Wischmeier and Smith, 1978, p. 4).

The USLE predicts long-term average annual soil loss on individual erosion plots. A major advantage of the USLE is that the equations are based on a large number of field studies under different environmental conditions and knowledge accumulated over several decades. The simple basic structure and the small number of input parameters required is one of the main reasons why the model was quickly distributed worldwide. As the above review shows great efforts were also been made in West Africa to gain knowledge from runoff-plot experiments to improve the factor values for the USLE such as rainfall-runoff erosivity, soil erodibility, and cover management values (Le Bissonnais et al., 1990; Diallo et al., 2004; Casenave and Valentin, 1989; Roose, 1977 and 1994). Nevertheless, for regions where no adequate datasets exist or where not enough data from runoff-plot experiments are available, the predictive capability of the model is low and uncertainties in factor values are high.

The USLE was originally based on and designed for small field scales, namely standard experimental plots of 22.1 m length and 9% slope gradient. With the advance of remote sensing and GIS techniques it was also applied to larger spatial scales such as catchments or regional scales (e.g., Kappas and Schweter, 1997; Lufafa, 2002; Ritchie, 2000). These applications beyond the specific modeling scale should be interpreted with care, because factor values at a field scale represent a specific process behavior inherent to this scale. This might be completely different at higher (or lower) scales. Due to limited soil loss data at larger scales, calibration and validation procedures are often neglected, but studies show that these investigations are meaningful and necessary. Hence, low prediction accuracy has been reported when simulated soil erosion rates were compared with actual soil erosion evidence from degraded fields (Cohen et al., 2006) or when soil loss simulated by the USLE was compared with results predicted by other models (Mati, 1999).

Some soil erosion researchers maintain that “there exists today no other environmental technology that is based on a larger and more comprehensive set of measurements”, and that the USLE predicts “average erosion rates reasonably well, even on recent, post-1960 conditions” (Nearing et al., 2000, p. 1300). Others point out that the “continued use of the unmodified USLE ignores fundamental problems” (Boardman, 2006, p. 77) and that, even from experimental results from runoff-plots, it is clear that the USLE is not satisfactory.

In order to overcome the limitations of the USLE, modifications and adjustments of parameter equations and nomographs have been produced to better reflect specific environmental conditions (Kinnell, 1995; Wang et al., 2002). The Revised Universal Soil Loss Equation (RUSLE), documented by Renard et al. (1997), retained the basic structure of the USLE, but incorporated some of these improvements. The RUSLE includes, for example, a revised R factor, an adjusted K factor, an improved LS factor, a supplemented P factor for tillage and residue management, and several subfactors, e.g., for canopy cover and soil moisture. As a successor of the USLE, the RUSLE was also intended to predict long-term average annual soil erosion on field plots and later additionally along hillslope transects. The model is used for soil erosion hazard mapping worldwide (e.g., Angima, 2003; Filippo et al., 2006; Märker, 2001; Millward and Mersey, 1999; Renschler et al., 1999), and its performance has been continuously improved by uncertainty analysis and accuracy assessments (Tran et al., 2002; Wang et al., 2001).

A major limitation of both models is the lacking representation of sediment transport and deposition. The model structure considers soil erosion as net soil loss over time and does not give any information about sediment yield, which might be important for off-site erosion effects. Although more recent studies promise to improve the RUSLE to predict off-site sediment accumulation (Biesemans et al., 2000; Foster, 2004), sediment yield is calculated based solely on interrill and rill erosion production. Other major source areas of sediment delivery such as concentrated flows, e.g., stream, channels or gullies, are neglected. Furthermore, both models are lumped and spatially not explicit. Soil loss is estimated at a point scale either from field plots or from sampling points along slope segments, and the final output is an averaged single soil loss value for the entire area. Also, the temporal component is neglected, as the model provides only averaged soil erosion rates over a long period of time - usually years - and does not account for seasonal variations in time that have, for example, an important effect on crop cover. Especially in tropical countries where crop cover varies highly due to dry and rainy seasons, soil erosion simulations might produce completely different results depending on the time of the year for which soil erosion is predicted.

Another often criticized limitation of the models is its oversimplified linearity, in which each environmental factor is presented by a single factor value that is then simply multiplied. This linear equation does not reflect any relationship between variables and can account neither for complex hydrological processes nor for the distinctiveness of individual environmental settings. As a result, a wide range of models has emerged that include a hydrological component in the modeling structure in order to simulate diffuse sediment discharge and off-site effects of soil erosion. Some examples of these models are ANSWERS (Areal Nonpoint Source Watershed Environment Response Simulation, Beasley and Huggins, 1982), AGNPS (Agricultural Nonpoint Source Pollution Model, Young et al., 1987) or CREAMS (Chemical, Runoff, and Erosion from Agricultural Management Systems, Knisel 1980), which all calculate event-based soil loss using a deterministic-analytical system approach. Although these models still include some parts of the USLE/ RUSLE environment, they can be seen as a first step toward physically-based models.

Physically-based erosion models calculate infiltration, runoff, erosion, transport and deposition processes in a deterministic-numerical way based on mathematical algorithms and physical laws of mass and energy flows. They have the advantage that they simulate the spatial and temporal distribution of soil loss and runoff more explicitly and thus might be more reliable for extrapolations to ungauged areas (Nearing and Nicks, 1998). Many physically-based models were developed to simulate soil erosion processes on different spatial and temporal scales, often with a specific scope of applicability and hence with a limited range of validity. For example, EUROSEM (European Soil Erosion Model; Morgan et al., 1992) or LISEM (Limburg Soil Erosion Model, De Roo et al., 1996) were developed to account for individual regional characteristics, whereas MEDALUS (Kirkby, 1998) and MEDRUSH (Abrahart et al., 1996) explicitly address the scaling issue.

The Water Erosion Prediction Project (WEPP) model (Flanagan and Nearing, 1995) is one of the most recent physically-based models, which was developed by the United States Department of Agriculture (USDA) as a “new generation prediction technology for use (…) in soil and water conservation and environmental planning and assessment” (Flanagan and Nearing, 1995, p. 1). The model can predict long-term as well as event-based soil erosion on hillslope or small catchment scales and allows a continuous simulation of vertically and horizontally distributed soil erosion and runoff processes. It differs from other models in that it partitions runoff between rill and interrill areas and calculates shear stress based on rill flow and rill hydraulics rather than sheet flow. The surface runoff is calculated using kinematic wave equations or an approximation method containing a regression equation for peak runoff rate and one for runoff duration. The hillslope erosion and deposition component is based on a steady-state sediment continuity equation to simulate the movement of suspended sediment in the flow with distance downslope. Effective hydraulic conductivity, a key parameter for the WEPP model, controls infiltration and runoff. Soil erodibility parameters are dominant parameters influencing the susceptibility or resistance of a soil to detachment and transport. The model also includes a stochastic climate generator (CLIGEN), an infiltration component based on the Green-Ampt approach, soil-water balance and percolation equations, a soil tillage and management component, and a plant growth component based on the phenological plant growth model of EPIC (Erosion Productivity Impact Calculator later renamed in Environmental Policy Integrated Climate, Williams et al., 1989).

The highly sophisticated, complex structure is typical for many physically-based models and illustrates the attempt to capture all fundamental natural processes that have an effect on soil erosion. The model is often a framework of well known physical equations and mathematical functions describing hydrological, pedological and biological processes on a micro scale and under very specific physical conditions. In order to simulate these processes in detail, a large number of heterogeneous parameters are required. The high demand of input parameters limits its applicability in countries for which no model-specific input database exists, such as tropical countries. Furthermore, highly detailed process description often leads to an over-parameterization of the model (Beven, 1989 and 1996). In the case that parameter values can be obtained from field investigations, measurement errors might occur, and the high spatial and temporal variability of environmental data will present one of the greatest constraints. If parameter values are not available, they must be extracted from observed data. Considering the large dataset and the possible high number of lacking parameters, the calibration procedure might involve high uncertainties in the identification of the correct value, and best fit solutions might go beyond acceptable values for individual parameters (Beck, 1987).

Furthermore, uncertainties might arise from the incompatibility between measuring and modeling scales. While most of the physical equations and algorithms are derived from investigations at smaller scales – often micro-plot or plot scale - and reflect physical process behavior at these scales (e.g Darcy`s law; Richards equation; Green-Ampt approach), they are often used in simulations at much greater scales, e.g., hillslope or catchment scale. By using point data to represent an entire grid cell, small-scale physics are lumped up to coarser areas, and the physical significance of equations becomes questionable (Beven, 1995; Seyfried and Wilcox, 1995). Considering these uncertainties in modeling, Beven (1996, p. 289) uses the term *equifinality* to express, “that good fits to the available data can be obtained with a wide variety of parameter sets that usually are dispersed throughout the parameter space.” Hence, equifinality implies uncertainties in inference and finally in predictions.

Another concern is that many models have been configured in a “piggyback fashion” (Rudra et al., 1998, p. 176), which implies that less important components have been piggybacked onto higher components. The result is a hierarchical structure in which errors in a fundamental component will lead to error propagation in a lower component although the basic values of the former might have been correct before. Additionally, the concepts of non-linearity, emergence and complex behavior are neglected in such approaches (Favis-Mortlock, 2004). Therefore, the assumption that physically-based models, which are designed to incorporate the laws of physics, are more reliable for extrapolations to ungauged areas (Nearing and Nicks, 1998) and could be applied beyond the range of conditions for which the model was validated for (Stone, 1995) should be reconsidered carefully. Despite the uncertainties involved, physically-based models tend to have a good explanatory depth and are useful to gain deeper knowledge about the process behavior of environmental systems – even though they are still far from being understood or simulated in all their details.

Both, physically and empirically based models have their specific advantages and limitations, and no model concept might be categorically superior over another or more appropriate in all situations (Table 2.2). The choice of the model will always depend on the guiding principle of parsimony, which says that “a parsimonious model is usually the one with the greatest explanation or prediction power and the least parameters or process complexity” (Mulligan and Wainwright, 2004, p. 8). But this causes another concern: while the explanatory depth of physically-based models is high, its predictive power is low, while the opposite holds true for empirically based models. Therefore, in terms of explaining processes and understanding the relationship between parameters, physically-based models are preferable.

On the other hand, however, their potential performance for accurate predictions is limited by the high degree of uncertainty associated with input parameters and finally with model outputs. Therefore, in terms of predictive power, simple empirical models might be more accurate, robust and reliable than highly complex models (Merritt, 2004; Perrin et al., 2001).

Although “the promised, ‘all-singing, all-dancing’ erosion models have not yet arrived” (Boardman, 2006, p. 73) there has been much progress made in soil erosion modeling during the last decade, especially in reconsidering the existing system approaches and theoretical concepts (Rhoads and Thorn, 1996c; Wainwright and Mulligan, 2004b). Additionally, a set of new models was developed taking into account the holistic behavior of natural systems (e.g., semi-quantitative models: De Vente and Poesen, 2005; Verstraeten et al., 2003) or integrating new technologies such as GIS and remote sensing into the modeling structure (e.g., distributed soil erosion models: Beven and Freer, 2001; Coulthard et al., 2000; De Roo, 1996; Jetten, 2003; Van Rompaey et al., 2001).

Table 2.2 Selected soil erosion models and their characteristics

Soil erosion model	Reference	Conceptual framework	Components	Temporal scale [(a)]	Spatial scale [(b)]	Major advantages	Major disadvantages
USLE (Universal Soil Loss Equation)	Wischmeier and Smith, 1978	Empirically based, static, lumped	Water erosion	1) long term 2) yearly	1) field plot 2) 1 element 3) not spatially explicit	Based on long-term expertise and many runoff plot data; low number of input parameters; easily applicable worldwide	No sediment deposition; not spatially/ temporally explicit; empirically based; over-simplification
RUSLE (Revised Universal Soil Loss Equation)	Renard et al., 1997	Empirically based, static, lumped	Water erosion	1) long term 2) yearly	1) field plot/ hillslope 2) 1 element 3) not spatially explicit	Based on USLE expertise (revised R and K factor, improved C and LS factor); low number of input parameters	No sediment deposition; not spatially/ temporally explicit; empirically based; over-simplification
WaTEM/ SEDEM (Water and Tillage Erosion Model; Sediment Delivery Model)	Van Oost et al., 2000; Van Rompaey et al., 2001	Empirically based, static, spatially distributed	Water erosion and sediment yield, tillage	1) long term 2) yearly	1) catchment 2) 1 element 3) finite raster elements	Sediment transport and delivery; topography driven; spatially explicit; raster based with gridcell-to-gridcell routing algorithm; high prediction power	Not temporally explicit; empirically based; over-simplification; low explanatory power
WEPP (Water Erosion Prediction Project)	Flanagan and Nearing, 1995	Physically and process-based, deterministic dynamic	Water erosion, runoff and sediment yield, hydrology, climate, tillage, management	1) long term event-based, 2) daily, event-based, breakpoint data	1) hillslope/ catchment 2) max. 8 elements 3) max. 10 elements	Spatial and temporal distribution of erosion and deposition; many climate and hydrological components and divers management options; high explanatory power	High input parameter requirements; large dataset; high degree of uncertainty; over-parameterization; low prediction power

[(a)] *1) period of simulation 2) time steps/ intervals for calculations* — *Based on Bork and Schröder, 1996*
[(b)] *1) dimension 2) vertical resolution 3) horizontal resolution*

2.2.4 Distributed models and GIS

The spatial representation of soil erosion processes in a landscape has been greatly facilitated by improved computing capabilities and advanced Geographic Information Systems (GIS). GIS techniques allow consideration of spatially explicit data by dividing the area into discrete units (e.g., raster cells, triangular irregular networks or irregular objects) for which soil erosion and deposition are simulated. A major advantage of the GIS approach is to reproduce the terrain of the landscape in more detail by integrating or embedding digital elevation/terrain models (DEM/DTM) into raster-based environmental/soil erosion models (Burrough, 1986; De Roo, 1998; Dikau and Saurer, 1999).

Therefore, in this study the analysis and visualization of spatial referenced information by GIS was a valuable tool for developing spatially distributed erosion models. The advantage of distributed models is that they account better for the local variability of soil erosion processes and land management decisions. These models represent spatially distributed soil erosion and deposition by water and often also reflect tillage processes in a two-dimensional landscape. Although most of them are still empirically based, they are topography-driven models and allow incorporation of the landscape structure by considering the spatial organization of different land units and the connectivity between them. They focus mainly on the spatial and less on the temporal variability of relevant parameters. Furthermore, they allow consideration of runoff and erosion dynamics in larger and more complex catchments and identify more accurately source and sink areas of sediment transport. By disaggregating the landscape into smaller functional units, mainly discrete grid cells in a uniform grid, distributed models calculate soil loss and runoff dynamics for each grid cell and finally accumulate predictions over the whole catchment (Pullar and Springer, 2000). The discretization of the landscape, and hence the optimal grid size, depends on the resolution, accuracy and availability of spatial information. If the precision of field data is low and the natural variability of the environment is high, the use of a high-resolution grid is limited, as insufficient information exists to represent spatially explicit environmental processes. Also, if input maps are interpolated and/or based on a wide range of estimates or subjective assumptions, they might easily lead to error propagation and amplification of uncertainties (Artan et al., 2000; Jetten et al., 1999). As the quality of prediction depends highly on the accuracy of spatial data, the scale of measurements should be closely related to the scale of basin discretization. Also the size and shape of spatial elements should be adjusted to those input parameters for which sufficient spatial information exists. Nevertheless, at the scale of the individual grid cell all models are lumped and based on lumped data (Merritt et al., 2003).

In regard to soil loss simulations, distributed models at catchment scales tend to produce higher uncertainties in result than field scale models. The main limitations are caused by the lack of spatial data, such as soil surface conditions, which leads to an increase in uncertainty and error propagation if the model is applied at larger scales. Therefore, calibration and model evaluation is a fundamental prerequisite for spatially explicit modeling (Boardman and Mortlock, 1998; Boardman, 2006). In cases where insufficient spatial data for a fully distributed model are available, the application of semi-distributed models is more appropriate. Semi-distributed models include elements of lumped models, as they divide the area into a cluster of landscape units, e.g., sub-catchments, for which the model is applied.

Examples for distributed or semi-distributed erosion models are ANSWERS (Beasley and Huggins, 1982), LISEM (De Roo et al., 1996), SEDEM (Van Rompaey et al., 2001), TOPMODEL (Beven and Frier, 2001) or WaTEM (Water Tillage Erosion Model, Van Oost et al., 2000). A new generation of more recently developed environmental models, e.g., agent-based models, follows the distributed approach and presents the dynamics and interaction of individual agents within a raster-based structure (Bithell et al., 2008) or Cellular Automata (Coulthard, 2002; Favis-Mortlock, 2004).

2.3 New paradigms: complexity, emergence and scale

Modeling of soil erosion processes requires a holistic understanding of the driving forces and underlying processes in a complex system. Although the literature review shows that many soil erosion studies exist that quantify the magnitude of runoff on a small scale (e.g., experimental plots), fewer studies assess soil erosion on a larger scale, i.e., on a landscape or regional scale. This might be due to the unsolved problems of dealing with different scales.

In recent years, many physically-based erosion models have been developed to quantify soil loss on a field or hillslope scale by considering the physical laws of energy and mass movement (see section 2.2.3). An effort has been made to gain a better understanding of small-scale processes (e.g., micro- and macro pores, surface crusting or seepage), which has often resulted in an extremely complex model structure suffering from over-parameterization, large datasets and complex computational requirements (Beven, 1989). However, neither feedback mechanisms nor non-linear responses or extrapolation of the models to ungauged/uncalibrated areas could be considered in these modeling approaches (Wainwright and Mulligan, 2004a).

This leads to the methodological question: Is the reductionistic approach the correct way of dealing with complex systems (Harrison, 1999)? Defining a system as a "set of objects or characteristics which are related to one another and operate together as a complex entity" (Summerfield, 1997, p. 9) means increasing our insight into the system by explaining its single components, but this detailed component analysis has obvious limitations, since the "whole is more than the sum of its parts" (Aristoteles c.330 BCE, cited by Favis-Mortlock, 2004, p. 355). Only recently, environmental scientists have attempted to incorporate this more complex view through concepts of emergence, self-organization and complexity in their system analysis (Harisson, 1999 and 2001; Favis-Mortlock, 2004; Phillips, 1999 and 2003). The emergent behavior of a complex system is a result of the interactions of its components in a non-apparent way (Bar-Yam, 1997; Wainwright and Mulligan, 2004a), but its integration in modeling approaches is still far from being solved (Favis-Mortlock, 2004).

These methodological reflections were also found to be essential for soil erosion research on different spatial scales in Burkina Faso, and therefore specific consideration was given in this study to the theoretical concept of scale.

2.3.1 Scale matters

The concept of scale in environmental science is more than just a simple measure for a cartographical unit (cartographic map scale) or an expression for the spatial size or dimension of an area (geographic scale). It refers above all to the *operational*, *observational* and *modeling scales*, which consider first the sphere of activity of natural processes, second the spatial or temporal resolution of objects affecting data collection, field measurements and sampling design, and third the working scale used to develop and apply environmental models (Zhang et al., 2004). From another viewpoint, these interpretations of scale can also be divided into methodological categories such as data, method, or model scale, and in phenomenological categories referring to the scale of processes, form, and storage (Diekrüger et al., 2007). These different definitions show that the concept of scale is not only dependent on the space and time, but also on the methods and approaches used to interpret the earth, and they show why the "the term 'scale' refers to a rough indication of the order of magnitude rather than to an accurate figure" (Blöschl and Sivapalan, 1995).

In recent years, scientists have become increasingly aware of the relevance and implications related to spatial and temporal scales, and consider scale-dependent relationships in their hydrological (Wood, 1995; Sivapalan, 2003), geomorphological (Dikau, 1989 and 2007; Schmidt and Andrew, 2005; Slaymaker, 2006) and soil erosion research (Kirkby, 1996; Peeters et al., 2008; Renschler and Harbor, 2002; Rickson, 2006).

By assessing soil erosion at different spatial scales, special emphasize is given to the operational time and space scales at which specific hydrological and pedological processes take place and which determine in return the selection of appropriate models to simulate these processes (Figure 2.4). For example, at smaller scales such as plot scales, soil properties related to vertical flows (e.g., infiltration capacity, drainage, soil texture, soil structure, aggregate stability) are most decisive for soil loss predictions. At hillslope scale, overland flow, shear stress and (inter-)rill hydraulics become important. At broader scales, such as watershed or basin scales, landscape properties and lateral soil and water flows (e.g., channel flow, gully erosion, lithology, geology, land-use, and long-term deposition in storages and reservoirs) become more relevant. Also, erosion controlling climate variables need to be adapted to the specific scale.

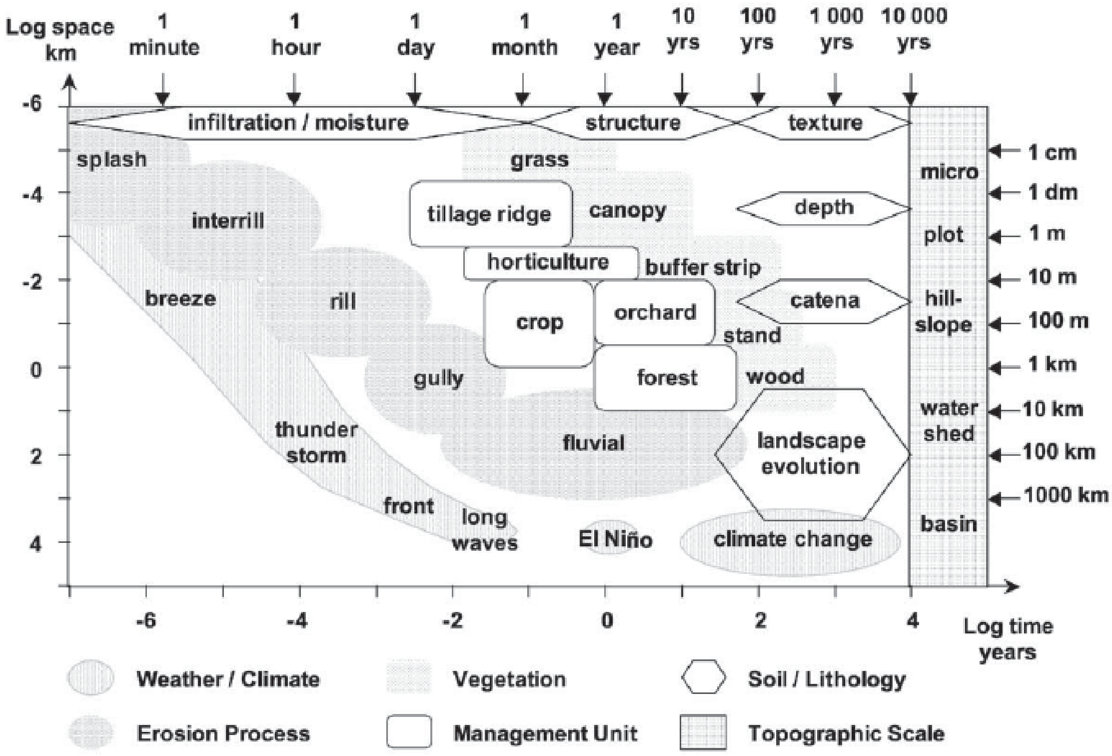


Figure 2.4 Space and time scale for erosion processes (Renschler and Harbor, 2002)

While the splash effect of raindrops might be the determining factor for the detachment of soil particles at a micro-scale, rainfall variability and climate change become dominant processes at larger and long-term scales. But the spatial pattern of environmental processes is also scale-dependent.

For example, the development of the channel network or the dynamics of land-use changes is not detectable at field scale because the spatial pattern is a result of synergism and emergence between individual system components at a higher scale of the system (Kirkby, 1998; Lambin, 2004; Albrecht and Car, 1999). Similarly, all anthropogenic factors and human decisions on land management and soil and water conservation practices refer to a specific scale. Whereas farmers usually tend to focus their decisions primarily on a field level, policy makers need to develop action plans or land-use and soil protection guidelines on a regional or national level. Moreover, land management and soil and water conservation techniques are more effective and sustainable if their spatial extent and temporal continuity are taken into account prior to implementation.

Besides these operational process scales, further scales need to be integrated for the environmental modeling approach: the modeling scale, the observational input or database scale, and the intended model output scale. Relevant scaling questions in this context are how to handle the spatial and temporal variability of natural processes, how to include different dominant processes at diverse scales, how to account for inherent process nonlinearities, how to simulate feedback mechanisms and cross-scale interactions, and how to reflect emergent properties and synergy effects of components.

Several methods have been developed to deal with these issues by so-called upscaling or downscaling approaches (Bierkens et al., 2000; McBratney 1998; Sivapalan, 2003; Zhang et al., 2004). Whereas upscaling is defined as transferring or extrapolating information from a lower to a higher scale, downscaling is associated with transferring or interpolating information from a higher to a lower scale (Bierkens et al., 2000). However, most techniques simply generalize or (dis-)aggregate parameter values, e.g., by using the averaging method, the thinning method or the dominant method, and hence do not solve the intrinsic scaling problem. In his hierarchical approach, Okoth (2002) scaled up lower level observations to higher levels by generalizing and smoothing parameter values. He found that the results obscured more than enhanced landscape system knowledge because the transfer of knowledge requires consideration of different attributes and drivers of soil erosion at each level. Gray (1999) concluded that processes detectable at one scale might not be detectable or relevant at another scale. By dealing with the question whether land is being degraded in Burkina Faso, he showed that this depends not only on the perception of the researcher or individual farmer but also on the scale of inquiry and the associated biophysical processes measured at that scale. Zhang et al. (2004) suggested first identifying all possibly scalable parameters and predominant, intrinsic processes at each scale, and then fitting both parameters and models to the required scale. He used the fractal method and the routing approach, which are more complicated but also more robust. Nevertheless, he raised concern about the fact that the reality of a variable is closely related to its corresponding process. Therefore, the closer a value is measured to its process scale, the more it can approximate reality.

However, direct upscaling or downscaling of processes and findings from one scale to another is not (yet) possible. Already Beven (1989) warned that severe problems in hydrological research are often related to the breaking of implicit scaling rules. Therefore, future research is required to discover synergy effects when downscaling or fragmenting parameter values and to account for emergent properties when upscaling or aggregating values (Slaymaker, 2006).

But still the question remains "How can we make a useful connection between the complexity observed at the hillslope scale and the apparent simplicity inferred at the watershed scale?" (Sivapalan, 2003, p. 1037). One possible option – albeit not solving the scaling problem – might be to follow a hierarchical or nested approach by focusing on multi-scale methods and multi-scale models to simulate soil erosion individually at each scale.

2.3.2 From hillslope to catchment scale

In this study, the scaling issue is accounted for by assessing soil erosion at field, hillslope, small reservoir, and catchment scale. The operational, observational and modeling scales are considered by collecting pedo-geomorphological, hydrological, climate and land-use/management data at the specific process scales, by considering the spatial and temporal resolution affecting the data collection and, finally, by simulating soil loss/deposition with two scale-dependent models.

In a total catchment area of approximately 1000 km², several smaller sub-catchments between 7 km² and 21 km² are selected, each characterized by a dammed reservoir (Figure 2.5). Within these sub-catchments, representative hillslopes of 200 m to 400 m length are chosen and, for a more detailed analysis, divided into hillslope positions characterized by single soil profiles. This hierarchical research design allows the collection of scale-dependent data and use of scale-adapted models to assess erosion dynamics from a smaller to a larger scale or from hillslope to catchment scale.

At the smallest research scale, here field or point scale, soil profiles are described along a catena, and soil horizon samples are analyzed for organic matter, bulk density, cation exchange capacity, texture, pH, carbon, nitrogen, phosphorus and potassium. These data provide valuable information about pedogenesis, soil-landscape development and nutrient status of the soil.

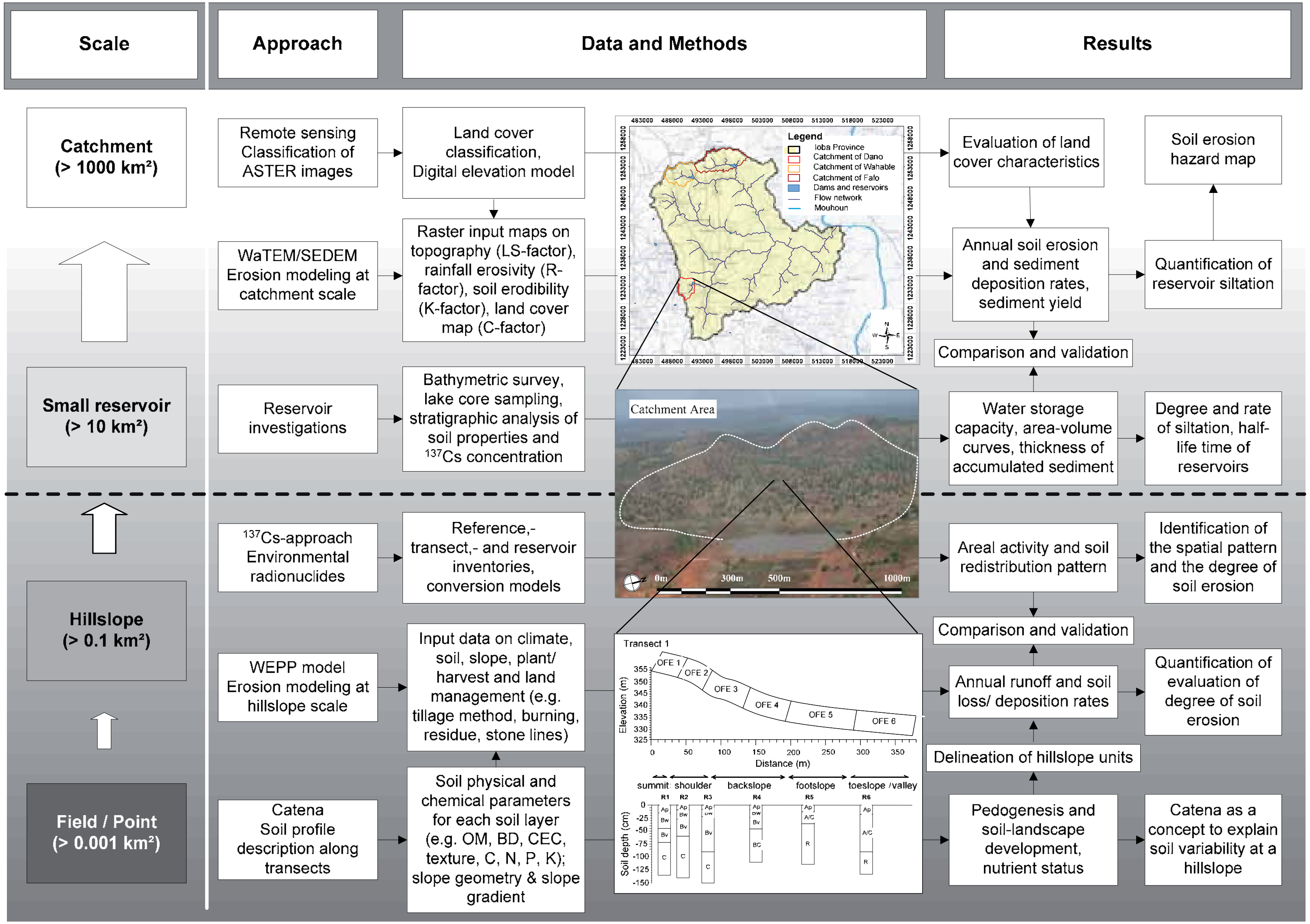


Figure 2.5 Research design for assessment of soil erosion and reservoir sedimentation at different spatial scale

Furthermore, the catena serves as a concept to explain soil variability along the hillslope and to assess systematic changes in soil properties affecting soil loss/deposition. To interlink these data with larger scales, the catenary approach is used to delineate the hillslope into individual units of pedo-geomorphic processes, which are reflected by the WEPP model as overland flow elements (OFEs).

At hillslope scale, the WEPP model is applied to predict annual soil loss/deposition rates and to simulate the effect of land management strategies on soil loss. A large dataset of climate, slope, soil and land management parameters is collected, while soil information is obtained from the previous soil profile analyses at field scale. The model allows prediction of soil loss/deposition rates for the entire hillslope as well as for individual hillslope positions. The ^{137}Cs approach serves as an additional method to identify the spatial pattern of soil erosion at hillslope scale. Reference, transect and reservoir inventories are taken and conversion models are used to convert the remaining ^{137}Cs concentrations in the soil into soil redistribution rates along the hillslope. The ^{137}Cs approach provides the option to compare measured soil loss/deposition rates with simulation results from the WEPP model.

At reservoir scale, processes related to sediment mobilization, sediment transport and sediment delivery become increasingly relevant. Therefore, off-site impacts of soil erosion, such as the accumulation of sediment in reservoirs, are assessed. A bathymetric survey is conducted to determine changes in reservoir bed morphology and to calculate losses in water storage capacity. Additionally, sediment cores of the reservoir bed are analyzed for stratigraphic changes in order to estimate the thickness of the accumulation layer and to calculate the magnitude of sediment yield. Besides downcore variations in soil properties, also variations in the ^{137}Cs concentrations are measured to test the suitability of the approaches and to determine the depth at which the in-situ, autochthonous soil of the reservoir bed is reached. Final results allow estimation of the degree and rate of reservoir siltation and prediction of the half-life of the reservoir.

The spatially distributed WaTEM/SEDEM model is selected as an appropriate model to simulate soil erosion and deposition rates at reservoir scale. For the input data files, raster-based maps on rainfall erosivity (R-factor), soil erodibility (K-factor), topography (LS-factor) and land-cover (C-factor) are required. The latter two maps are based on extracted maps from remote sensing images providing spatial information for the larger catchment area of 1000 km². Whereas the DEM is derived from SRTM-data and downscaled to a higher resolution, the land-cover map is based on an (un)supervised classification from ASTER-images. The WaTEM/SEDEM model generates output maps of the spatial distribution of soil loss/deposition rates and calculates the total amount of sediment production, sediment deposition, sediment/river export and pond deposition. These results can be compared with the measured results from reservoir investigations to provide valuable estimates on reservoir siltation. Furthermore, the simulated output maps can be used to define soil loss tolerance values and to identify environmental hazard zones at larger scales.

This research design allows the provision of knowledge at a wide range of scales, accounts for the dominant erosion and deposition processes at each scale individually and, finally, uses the advantages of multiple techniques to compare and validate soil erosion and reservoir sedimentation results within and between scales.

3 SOIL EROSION ASSESSMENT AT HILLSLOPE SCALE

3.1 Introduction

While numbers about the exact amount of soil loss might vary highly depending on environmental conditions and methods used for its quantification, scientists, policy makers, extension workers and, above all, farmers agree that soil erosion is a serious environmental problem in Burkina Faso (Gray, 1999; Hessel et al., 2009; Roose, 1994; Sterk et al., 2005; Visser, 2003).

For the assessment of soil erosion, a hillslope represents an adequate process scale for analyzing the dynamic interaction of vertical and lateral flows of soil-water in detail without neglecting its impact for a broader, complex landscape (Braid, 2004; Kirkby, 1998; Sivapalan, 2003). Although smaller scales, such as experimental plots, might be more appropriate to gain insight into micro-scale processes (e.g., infiltration capacity, erodibility, aggregate stability, cohesion characteristics or micro- and macropores), they can neither consider lateral flows between fields nor take soil redistribution patterns along slopes into account and may often lead to misleading results (Boardman, 2006). Additionally, in contrast to experimental plots or fields, the hillslope can be seen as a distinct landscape unit representing a differentiable geomorphological surface form that is part of a larger landscape system (Carson and Kirkby, 1972; Dikau, 1989; Schmidt and Andrew, 2005; Wilson and Galant, 2000).

When considering the appropriate scale for soil erosion assessment, field boundaries and process integration within a landscape system might also be important especially in regard to soil and water conservation techniques as well as land management strategies. These often rely on soil loss estimates to allocate resources and to select priority areas for development programs and action plans (Gobin, 1999; Stroosnijder, 2005). Soil erosion patterns and redistribution rates might be preferable to single numbers for identifying erosion-susceptible areas more precisely and for adjusting soil conservation and land management methods to the specific needs of degraded soils.

In terms of erosion susceptibility and soil variability, most studies differentiate only between upland and lowland soils and propose the adoption of farming systems to these two categories (Mulders et al., 2001; Stoop, 1987). Even if a delineation into these main categories reflects the strongest contrast in terms of soil types and nutrient status as well as a high variability in pedo-geomorphological, hydrological and physio-chemical soil properties, it neglects significant transitions between these extremes, which might considerably influence the interpretation of soil erosion results.

The catena concept was chosen as the theoretical framework to account for the systematic changes in pedo-geomorphological and hydrological properties along the slope, which influence soil erosion susceptibility, sediment transport and soil redistribution patterns. The catenary succession of soils assumes that each point on a slope has distinct pedogenic characteristics reflecting soil-water-gravity interrelations determined by its surface form (Conacher and Darymple, 1977; Milne, 1935; Ruhe, 1960). Based on these soil-water-gravity interactions, the hillslope can be further delineated into individual surface units, so-called process-respond units, which possess specific pedo-geomorphological characteristics for each individual hillslope section. Such a systematic delineation also makes it possible to account for the spatial variability of soils within a larger landscape system, which has been seen to be an advantage in many process studies (Hall and Olson, 1991; Kirkby et al., 1996; Park et al., 2001). Moreover, recent modeling developments emphasize the importance of including such spatial heterogeneity of soils and landform morphology in modeling approaches for assessing and interpreting the spatial distinction of erosion and deposition processes over the landscape (Brunner et al., 2004; Märker, 2001; Mitas and Mitasova, 1998; Papiernik et al., 2009; Park and Vlek, 2002; Van Oost et al., 2000).

In this study, the suitability of the catena concept for soils in southwestern Burkina Faso is first examined by soil profile investigations and soil horizon analyses of four transects at two different research sites. The specific objectives are i) to assess catenary soil development along contrasting hillslope transects, and ii) to evaluate variations of physical and chemical soil properties along these transects.

Secondly, an environmental tracer method (^{137}Cs) is applied to assess the severity of soil erosion and to identify the most erosion-affected hillslope zones. The ^{137}Cs technique is a more recent, but promising method to quantify soil erosion and deposition over the landscape by measuring the remaining concentration of ^{137}Cs in the soil (Mabit et al., 2008; Schumacher et al., 2005; van Oost, 2003; Walling and He, 2001; Walling and Quine, 1993; Zapata, 2002). This technique can provide mid-term erosion rates (over approximately 40-50 years) and presents a possibility to obtain information about the quantity of soil that is eroded, re-distributed and finally accumulated downslope. The isotope ^{137}Cs was produced by radionuclear weapon testing during the 1960s, where ^{137}Cs was distributed globally within the stratosphere and deposited worldwide as radionuclide fallout. ^{137}Cs, which is not produced naturally, has a half life of 30.17 years, indicating that today approximately 30-40 % of the original radionuclide fallout still remains in the soil (Walling and Quine, 1993). Today, only a few studies are known that have used the ^{137}Cs method in West Africa, but these studies have shown that the radionuclide technique might be appropriate for soil erosion assessment in semi-arid environments and could therefore present an effective tool for validating soil erosion models (Chappell et al., 1998; Faleh et al., 2005; Pennock, 2000). The specific objectives of the ^{137}Cs approach in this study are i) to quantify the amount and the range of soil loss and soil deposition on two hillslopes, and ii) to evaluate the soil redistribution pattern of ^{137}Cs derived from different conversion models (proportional model, mass balance model 1, 2 and 3).

Finally, a soil erosion model was applied to simulate soil loss rates at hillslope scale by considering slope-specific pedo-geomorphological characteristics and site-adequate land management options. Here, the physically-based Water Erosion Prediction Project (WEPP) model (Flanagan and Nearing, 1995) was selected to predict both long-term and event-based soil loss along transect lines and to account for horizontally distributed soil erosion processes. Although the WEPP model was first developed for environmental conditions in the Unites States, it can be adapted to many different conditions and has been applied worldwide. The WEPP model was also chosen as it allows consideration of the catena concept by dividing the hillslope into several subunits reflecting systematic variations in soil properties. The model covers a wide range of applications for cropland and has been applied to simulate the effect of various farming strategies, such as tillage (Schumacher et al., 1999; Yu et al., 2000; Zhang, 2005) and soil conservation strategies (Brunner et al., 2008; Gilley and Doran, 1998; Shen et al., 2010; Zhou et al., 2009).

The specific objectives of the soil erosion modeling approach are i) to simulate soil loss and soil deposition rates on two hillslope transects by considering slope-specific variations in soil properties based on the catena concept, ii) to predict the effect of soil and water conservation as well as land management techniques (e.g., minimum tillage, residue management, and contour farming) on simulated soil loss in order to identify potential land management strategies to counteract soil erosion and iii) to compare simulated soil loss values with ^{137}Cs estimates along a transect line (derived from mass balance model 3).

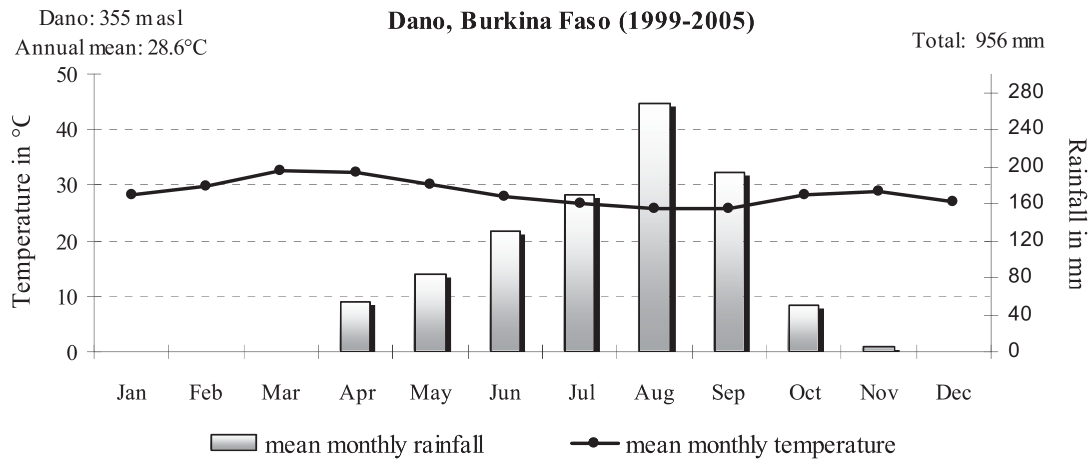
3.2 Site description

For comparative studies, two representative hillslopes were selected from different sites within the Ioba Catchment, southwestern Burkina Faso (Table 3.1):

1) A hillslope within the subcatchment of Dano, located at latitude 11°15’ N, longitude -3°09’ W and at 355 m asl draining into the reservoir behind the Dano-dam (“Barrage du Moutouri”),

2) A hillslope within the subcatchment of Wahable, located at latitude 11°32’ N, longitude -3°08’ W and at 328 m asl draining into the Wahable-dam reservoir (“Barrage du Wahable”).

Both study sites have an average annual precipitation of approximately 960 m and belong to the transition zone between the Sudano-Sahelian zone in the north and the Sudanian zone in the south. The 900 m isohyet is the limit between these two climate zones. The rainfall pattern is mono-modal with a rainy season from May to October and a dry season from November to April (Figure 3.1).



Rainfall data (1999-2005): DPAHRH/ Ioba, Direction régionale de l'Agriculture de l'Hydraulique et des Ressources Halieutiques du Sued-Ouest, Burkina Faso. Temperature data (2006): Climate station in Dano, Project Glowa-Volta, ZEF, University of Bonn.

Figure 3.1 Average monthly temperature and rainfall pattern in Dano, Burkina Faso

Table 3.1 Environmental characteristic of study sites

General category	Specific data for Dano	Specific data for Wahable
Climate [a]	• Climate zone: Sudano-Sahelian • Average annual rainfall: 956 mm • Rainy season: May until October • Dry season: October until April • Average annual temperature: 28.6°C • Years of high rainfall (>1200mm): 1951, 1962, 1963, 2003 • Years of low rainfall (<800mm): 1973, 1983, 1984, 1990, 2005	• Climate zone: Sudano-Sahelian • Average annual rainfall: 956 mm • Rainy season: May until October • Dry season: October until April • Average annual temperature: 28.6°C • Years of high rainfall (>1200mm): 1951, 1962, 1963, 2003 • Years of low rainfall (<800mm): 1973, 1983, 1984, 1990, 2005
Geology/ Geo-morphology	• Ioba Mountains :Birimian mountain range (Middle Precambrian) • Magmatic rocks: Dolérites (Gabbro) and Andésites. Metamorphic rocks: quartzite • Strong physical and chemical weathering processes • Slight undulating surface form with concave and convex slopes • Seasonal overland flow during rainy season; formation of runoff rills	• Birimian peneplain (Middle Precambrian) • Crystalline, metamorphic rocks and small lateritic hilltops • Strong physical and chemical weathering processes • Almost level surface form with predominantly linear slopes • Seasonal overland flow during rainy season, formation of runoff rills
Soils	• Rhodic cambisols, clay to silty clay, richness of bases, good soil fertility, favorable for agriculture • Leptosols, clay loam to loam texture, very shallow topsoil, limited water holding capacity	• Ferric lixisols, sandy loam to clay loam, low organic matter, chemically depleted/ leached, low soil fertility, less favorable for agriculture • Stagnic lixisols (valley/ footslopes)
Vegetation/ Land-use	• Vegetation zone: Sudanian domain • Natural vegetation: Wooded, arboraceous or scrubby savannah, abundant perennial grasses • Land-use since 1950s: continuously annual crops, mainly sorghum, millet • Tillage practices: handhoe • Soil conservation practices: stone lines • Residue management: soil cover with millet/ sorghum stems after harvest	• Vegetation zone: Sudanian domain • Natural vegetation: Wooded, arboraceous or scrubby savannah, abundant perennial grasses • Land-use since 1950s: continuously annual crops, mainly sorghum, millet • Tillage practices: ox-plough • Soil conservation practices: contour ploughing • Residue management: soil cover with millet/ sorghum stems after harvest

[a] *DPAHRH/ Ioba, Direction régionale de l'Agriculture de l'Hydraulique et des Ressources Halieutiques, du Sued-Ouest; Precipitation data from 1999-2005*

The onset of the rainy season is highly variable and starts -depending on the year - mid/end of May. The first scattered rainfalls in April belong still to the dry season (the so-called "Mango-rain") and indicate the end of the dry season. During the rainy season, monthly rainfall increases to max. 270 mm in August, at the peak of the rainy season, with a relative humidity of 75 %. During the dry season, the relative humidity is less than 20 % and the Harmattan wind, a weather phenomenon caused by the oscillation of the Inter Tropical Convergence Zone (ITCZ), brings dust-laden air masses with high aerosol concentration from the Sahara, reaching its maximum extent in January. Whereas the minimum, monthly temperature is 20.1°C in December, a maximum monthly temperature of 38.4°C can be reached in March.

The natural vegetation is part of the Sudanian domain with wooded, arboraceous or scrubby savannah and often abundant perennial grasses. However, the primary vegetation is disappearing due to human influence and agricultural activities, which are changing the vegetation pattern more and more into a man-made cultivated landscape predominated by annual crops, such as millet (*Pennisetum glaucum*), sorghum (*Sorghum bicolor*), maize (*Zea mays*) or cowpeas (*Vigna unguiculata*).

The relief of Dano is formed by the Ioba Mountains, the remnants of an ancient eroded Birimian mountain range with NNE-SSW orientation, which is from mid-Precambrian approximately 540 to 1800 million years old. The Birimian chain is composed of magmatic rocks, predominantly Dolérites (Gabbro) and Andésite. Also metamorphic rocks, such as quartzite and its mineralization forms like pyrite and silexite (quartzolite) can be found which indicate strong physical and chemical weathering processes - typical for this climatic zone. The predominant soil type is a *rhodic cambisols* (WRB, 1998) which is equivalent to *sol brun eutrophe peu évolué faciès ferruginisé* (CPCS, 1967) in the French classification scheme. Characteristic is their fine clay to silty clay texture, their illuviation horizon and often the occurrence of a plinthite layer in the lower parts of the soil profile. Due to their high cation exchange capacity and good quality and richness in bases, these soils are favorable for agriculture and are used to cultivate annual crops like millet, sorghum and cowpeas. In terms of terrain attributes, the hillslope of Dano has a summit-valley distance of approximately 350 m and a lateral width of approximately 200 m. Average slope gradients are 4°, varying from a maximum of 25° at shoulder/backslope positions to 0-1° at footslope/valley positions. Concerning the landforms, the hillslope can be characterized morphologically as a slight undulating surface form, which leads to converging water and sediment flows on concave slope positions and diverging flows in convex positions.

The second hillslope, Wahable, is located at a distance of approximately 20 km from Dano and shows distinguished pedological and morphological characteristics, which might be seen as more representative for the Ioba province in general. The landscape is also part of the Birimian formation, but the relief is only weakly accentuated. The ancient, leveled peneplain consists of crystalline, metamorphic rocks and small lateritic hilltops, which have a height difference of 50-80 m compared to the average altitude of 300 m of the surrounding monotonous relief. The soils are predominantly *ferric lixisols* (WRB, 1998) which are equivalent to *sol ferrugineux tropicaux lessivé* (CPCS, 1967). These highly weathered tropical soils have a chemically depleted eluvial horizon and a sandy loam to sandy clay loam texture. The high erosion susceptibility, low pH value and low organic matter content restrict its use for agriculture. Nevertheless, millet, sorghum and maize are cultivated on these soils. The hillslope in Wahable has a summit-valley distance of approximately 240 m and a lateral width of approximately 200 m. The almost leveled surface has a low average slope gradient of 2.6°, ranging from 10° at backslope to <1° at valley positions. Therefore, no significant redistribution of water and sediment is expected, except in the case of intensive or continued rainfall events, which occur especially in the transition time from the dry to rainy season.

3.3 Data and methods

3.3.1 Topographical survey

The Digital Elevation Model (DEM) for the hillslope in Dano was based on field measurements using a tachymeter (Leica, type Wild NK-1 non-automatic tilting level). In a spatial grid of 10 m by 10 m, a total of 574 points were positioned. All elevation points were processed by point kriging in Surfer software (Golden Software, version 8.0). The accuracy of the DEM was 0.05 m for elevation.

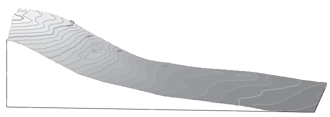
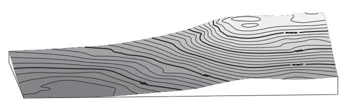
A Differential Global Positioning System (DGPS) from Ashtech (Promark 3) was used to generate the DEM for the Wahable hillslope. A known 3-D control point was selected as reference for the base station. A total of 240 GPS points were collected on an almost level surface by a mobile antenna (stop-and-go survey) in a sampling grid of approximately 20 m by 20 m. Data processing with GNSS-Solution software from

Ashtech showed an average accuracy of 0.04 m for elevation. The DEM was generated by point kriging in Surfer.

3.3.2 Terrain analysis

Terrain attributes were derived from the DEMs by DiGeM 2.0 (Conrad, 2002) (Table 3.2). Morphometric properties, such as slope, aspect, plan and profile curvature were based on the method of Zevenbergen & Thorne (1986).

Table 3.2 Terrain attributes and indices derived from digital elevation models

Attribute	**Hillslope of Dano**	**Hillslope of Wahable**
Elevation (m asl.)	Max: 356 m, min: 326 m	Max: 330 m, min: 324 m
Relief (m)	30 m	6 m
Aspect (degrees)	Southeast 112°	South/ Southwest 194°
Slope gradient (degrees)	Average: 4.1°, max: 25.4°, min: 0.7°	Average: 2.6°, max: 9.7°, min: 0.0°
Slope length (m)	350 m	240 m
Profile/Plan curvature	Convex/linear (shoulder) Straight/linear (backslope) Concave/linear (footslope) 	Convex/linear (shoulder) Straight/linear (backslope) Concave/linear (footslope) 
Wetness index (WI= ln (As/tanß)[a]	Average: 5.3, max: 16 (footslope/valley), min: 2 (shoulder)	Average: 6.8, max: 18 (footslope/valley), min: 3 (shoulder)
Stream power index (SI = As*tanß)[b]	Average: 5.4, max: 235, min: 0	Average: 1.0, max: 40, min: 0

[a] WI= wetness index, [b] SI = Stream poxer index, As= specific catchment area, ß = slope gradient

The Deterministic 8 method by O'Callaghan and Mark (1984) was selected to generate flow accumulation, because this one-dimensional model produced the most realistic flow accumulation pattern for both hillslopes compared with the two-dimensional Multiple-Flow Direction (MFD) methods by Quinn et al. (1991) and Freeman (1991), which showed a high scattering. Topographic parameters, such as wetness index and stream power index were derived based on the standard algorithms by Moore et al. (1992).

3.3.3 Soil sampling and laboratory methods

On each hillslope, two representative catenary transects were selected extending parallel from summit to footslope/valley position. In Dano, the transects had a total summit-valley length of 350 m and a distance of 40-60 m between them. Along each transect 6 soil profile pits were dug with a dimension of 1x1 m and a depth of at least 1.5 m. The spacing between the single soil profiles was 30-80 m (Figure 3.2). In Wahable, the summit-valley length of the transects was 240 m and the distance between them 50 m. Along each transect 5 soil profile pits were dug, likewise with a dimension of 1x1 m and a depth of at least 1.5 m. The spacing between single soil profiles was 40-60 m. (Figure 3.3).

Soil horizon description was based on the standardized FAO field description (FAO, 1977 and FAO, 1994), soil classification was based on the French classification scheme (CPCS, 1967) and the international World Reference Base for Soil Resources (WRB, 1998 and 2006). Soil composite samples were taken per each horizon by a standardized hammering head (100 cm^3) and analyzed for organic matter, bulk density, texture, pH, carbon (C total), nitrogen (N total), phosphorus (P available), potassium (K available), cation exchange capacity (CEC_{pot}) and exchangeable cations.

For laboratory analysis, soil samples were air-/oven dried, weighed and sieved through a 2 mm sieve. Bulk density was calculated by the ratio of dry mass to volume (WRB, 2006). Soil texture was determined using the Köhn-Pipette analysis (DIN 19683, 1973). The pH was potentiometrically measured in a 1:2.5 KCl solution. Organic matter was determined by the loss-on-ignition method (VDLUFA Methodenhandbuch, 1999) and for clay soils by multiplying total carbon (C_{org}) y the factor two. Total carbon and total nitrogen were determined by dry combustion in a C/N analyzer (Euro Vector CHNS-0 Elemental Analyzer; Model: EuroEA 3000; Serie: Euro Vector SpA). Available potassium and available phosphorus were analyzed using the CAL-method (VDLUFA Methodenhandbuch, 1999). Potential cation exchange capacity (CEC_{pot}) and exchangeable cations (K^+, Na^+, Ca^{++}, Mg^{++}) were extracted with $BaCl_2$-TEA at pH 8.2 (modified from Mehlich, 1953). In contrast to the effective CEC (CECeff), which determines soils actual exchange capacity at the current pH value, the potential CEC is a measure of soils maximum capacity to adsorb and exchange cations at a fixed, neutral pH value (Scheffer and Schachtschabel, 2002).

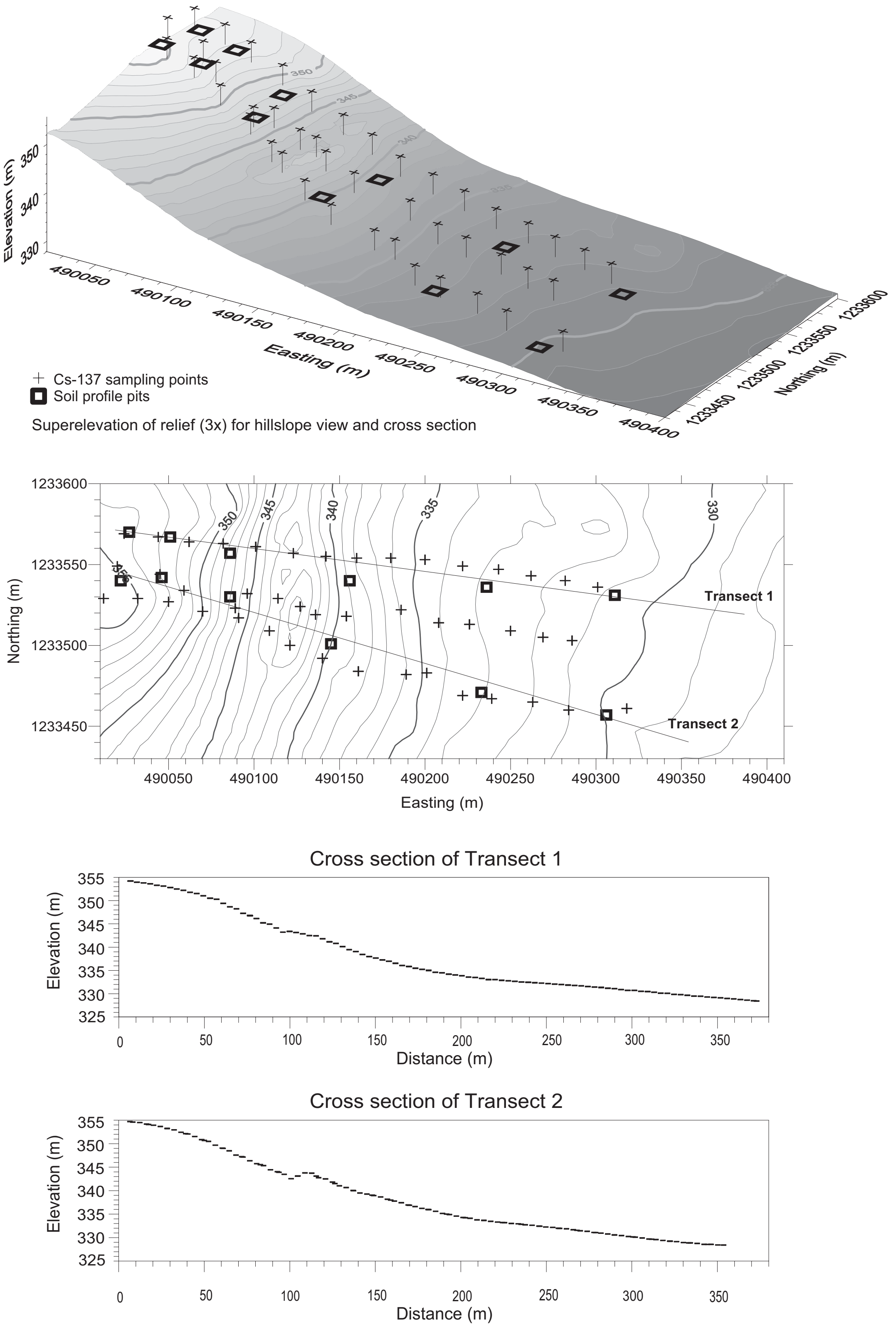


Figure 3.2 Digital elevation model, contour map and cross section of transects on hillslope in Dano, Burkina Faso

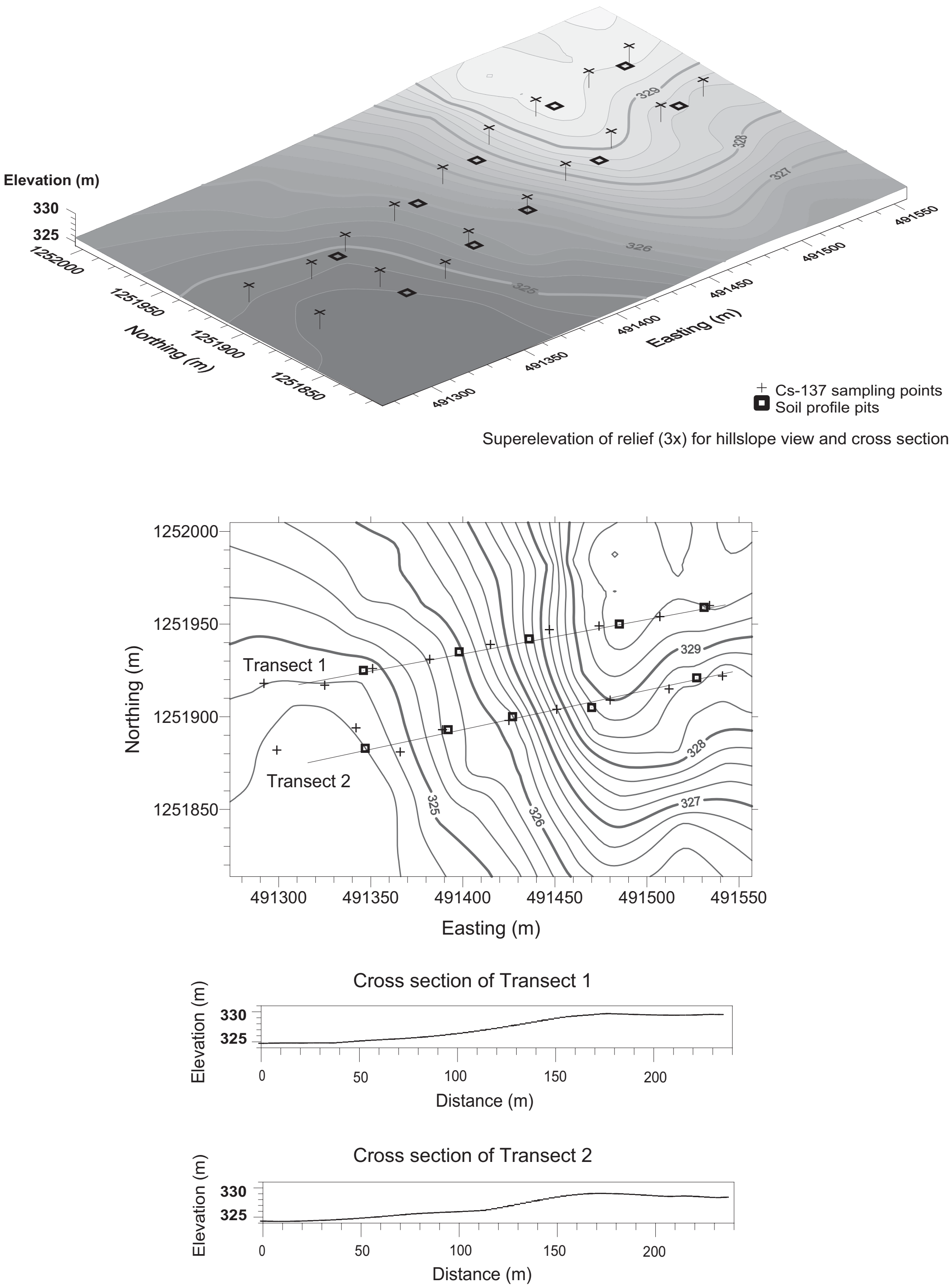


Figure 3.3 Digital elevation model, contour map and cross section of transects on hillslope in Wahable, Burkina Faso

3.3.4 ^{137}Caesium sampling and processing

Reference samples were taken from undisturbed sites to provide an estimate of total ^{137}Cs fallout since the mid 1950s for the Ioba province. The half-life of ^{137}Cs is 30.17 years, which indicates that approximately 30-40 % of the original radionuclide fallout still remains at undisturbed sites. Three reference sites could be identified on plateaus or flat areas, which were not cultivated during the last 50 years (based on farmers' knowledge). They were neither influenced by erosion nor deposition and were covered by grassland. Out of the three reference sites, only one reference site in the Natural Reserve of Bontioli about 30 km south of the study sites proved to be an adequate undisturbed reference site regarding the depth profile of ^{137}Cs activity. The total ^{137}Cs inventory was 729 Bq m^{-2}. The Natural Reserve of Bontioli was established in 1957 and is since then a protected area for many decades and has a natural savannah vegetation and no agricultural activities. Within this natural reserve, a large flatplain covered with grass and shrubs was selected for sampling.

Five composite samples at a distance of 10-15 m were combined to reduce the spatial variability of individual sampling points. The bulk composites were taken in 5 cm depth increments until a depth of 30 cm. A deeper depth was not possible, because all sites showed an impermeable hard rock layer starting 30-35 cm below the surface, probably also the reason for the abandonment of this land for cultivation. In order to obtain insight on ^{137}Cs distribution with deeper depth, a soil sample of 5 cm depth increments until a soil depth of 60 cm was taken from the dry reservoir of Dano. At this depositional site, the ^{137}Cs concentration at a depth of 30-35 cm was up to 0.5 Bq kg^{-1} with a measurement error of above 33 % (compared to 1.1 Bq kg^{-1} and an error of 16 % at 25-30 cm), which decreased to 0.2 Bq kg^{-1} with a measurement error of 61 % at a depth of 35-40 cm. These very low ^{137}Cs concentrations below a depth of 30 cm are seen as not reliably detectable. They can rather be explained by movement of fine soil particles within the soil than by initial ^{137}Cs fallout.

For erosion assessment at hillslope scale, 46 samples were collected at the hillslope in Dano in a grid sampling design composed of three parallel ^{137}Cs transects from summit to footslope/valley positions. Each transect contained 15 to 16 sampling points approximately 20 m apart from each other (Figure 3.2). The distance between the transects was approximately 40 m. At each sampling point, two bulk composites were taken approximately 3-4 m apart. For the almost level site in Wahable, 18 samples were collected along two parallel transects. The spacing between the sampling points along each transect was 30 m, the spacing between transects about 50 m (Figure 3.3). At each sampling point, three bulk composites were taken within a triangular distance of 3-4 m. All samples were collected by an undisturbed soil sampler (Split Tube Sampler, Eijkelkamp), which had a diameter of 48 mm, a surface area of 18,1 cm² and a volume of 543 cm^3 considering a soil depth of 30 cm. This maximum sampling depth was based on findings of reference inventory samples, which showed no significant radioactivity of ^{137}Cs below a soil depth of 30 cm.

The samples were disaggregated by hand and oven dried at 105° until a constant weight was achieved. The rock fragments (>2mm) were separated from the fine fraction (<2mm) using a 2-mm mesh sieve and weighed. Bulk density was calculated by the ratio of dry mass to volume (WRB, 2006). A subsample of 300g was analyzed by gamma spectroscopy in the isotope laboratory (ISOLAB) of the Department of Physical Chemistry at the University of Göttingen, Germany. The radioactivity of ^{137}Cs in the soil was measured by a HPGe detector of 26 % relative efficiency using a measuring time of 250.000 seconds to reduce the measuring error below 10 %. The gamma peak at 662 keV indicated the presence of ^{137}Cs in the sample. The peak count information was processed and stored by a multi-spectral analyzer (MCA) and converted into specific activities (Bq kg^{-1}).

Due to the high percentage of rockfragments, especially in the soils from Dano (average percentage of rockfragments >37 %), the bulk density of the soil was corrected for the influence of concretions as suggested by Auerswald and Schimmack (2000) and described by Pennock and Appleby (2002). The areal activity (Bq m^{-2}), a measure of the total amount of ^{137}Cs per unit area, was calculated following the equation of Sutherland (1994) in order to take the corrected bulk density for the fine fraction into account:

$$^{137}Cs \;\; inventory = \sum_{i=1}^{n} C_i \times BD_i \times DI_i \times 1000 \tag{3.1}$$

where ^{137}Cs inventory = total areal inventory (Bq m^{-2}), i = sampling depth, n = maximum number of sample depth with detectable ^{137}Cs, C_i = activity concentration per unit mass (Bq kg^{-1}) for depth i, BD_i = dry bulk density (Mg m^{-3}) for depth i, and D_i = depth increment (m) for sample i (Sutherland, 1994, p.61-62).

3.3.5 ^{137}Caesium conversion models

The proportional model and the mass-balance models 1 and 2 were used to convert ^{137}Cs inventories (Bq m^{-2}) to redistribution rates (t ha^{-1}) for each sampling point and the mass-balance model 3 was used for ^{137}Cs rates along transects, as shown below. The point estimates were aggregated to the entire hillslope by point-kridging in Surfer (Golden Software, version 8.0). This procedure allowed firstly to compare the redistribution pattern of ^{137}Cs derived from each conversion model, secondly to calculate the total and average amount of erosion and deposition for each zone separately and finally to identify the sediment delivery ratio for the hillslope.

The proportional model presents a linear function in which ^{137}Cs redistribution rates are calculated based on differences between ^{137}Cs reference inventory and Cs^{137} concentrations on individual sampling points, actual plough depth and bulk density of the soil and the time elapsed since initiation of ^{137}Cs accumulation by fallout. If an individual sampling point has a lower ^{137}Cs concentration than the local reference ^{137}Cs inventory, soil is eroded from that location, and if the ^{137}Cs concentration is higher, sediment is accumulated. The equation can be expressed by:

$$Y = 10\frac{BdX}{100TP} \tag{3.2}$$

where Y = mean annual soil loss (t ha^{-1} yr^{-1}), d = depth of plough or cultivation layer (m), B = bulk density of soil (kg m^{-3}), X = percentage reduction in total ^{137}Cs inventory (defined as $(A_{ref}-A)/A_{ref}\times 100$), T = time elapsed since initiation of ^{137}Cs accumulation (yr), A_{ref} = local ^{137}Cs reference inventory (Bq m^{-2}), A = measured total ^{137}Cs inventory at the sampling point (Bq m^{-2}), P = particle size correction factor (Walling and He, 2001, p. 6).

More sophisticated mass balance models were developed by Kachanoski and DeJong (1984) and further improved by Walling and Quine (1991) and Quine (1995) in order to take the effect of initial fallout distribution, its temporal variation, tillage translocation, seasonal timing of rainfall erosivity and particle size selectivity on ^{137}Cs redistribution rates into account. The general framework of the mass balance models for an eroding site can be represented as:

$$\frac{dA(t)}{dt} = (1-\Gamma)I(t) - (\lambda + P\frac{R}{d})A(t) \quad (3.3)$$

where A(t) = cumulative ^{137}Cs activity per unit area (Bq m^{-2}), R = erosion rate (kg m^{-2} yr^{-1}), d = cumulative mass depth representing the average plough depth (kg m^{-2}), λ = decay constant for ^{137}Cs (yr^{-1}), I(t) = annual ^{137}Cs deposition flux (Bq m^{-2} yr^{-1}), Γ = percentage of the freshly deposited ^{137}Cs fallout removed by erosion before being mixed into the plough layer and P = particle size correction factor (Walling and He, 2001, p. 8).

The required input parameters for the conversion models were adjusted to soil and land management conditions in Burkina Faso and either obtained from field measurements or based on default or recommended literature values under similar conditions (Table 3.3).

Information about the temporal distribution of ^{137}Cs-fallout for the northern hemisphere between 1954 and 1988 was provided by a dataset included in the model-software package. Chernobyl-derived fallout in 1986 need not to be considered for West Africa, as the Chernobyl plume did not reach the stratosphere and fallout then was limited to Europe and the former USSR (De Cort et al., 1998; Golosov, 2002; Walling and Quine, 1993).

A reference inventory A_{ref} of 729 Bq m^{-2} was considered as adequate to present ^{137}Cs-fallout at an undisturbed site. Tillage depth was assumed to be 0.10 cm to reflect conditions for hand hoe ploughing in Dano and for ox-plough tillage in Wabable, where farmers had recently started contour ploughing using animal power. The proportion factor was set to 0.5 to indicate that a high proportion of erosive rainfall occurs before the cultivation period and hence leads to a partial removal of ^{137}Cs before it is completely incorporated into the plough layer. The relaxation depth, a measure for the depth to which the initial ^{137}Cs-fallout infiltrates into the soil, was set to 4.00, a recommended value for cultivated soils (Walling et al. 2002). The tillage constant depends on tillage operation and timing and was assumed to be 200 for hand-hoe tillage in Dano and 250 for ox-plough tillage in Wahable. These values were based on the tillage constant estimates by Quine et al. (1997), who suggested values between 250 and 350 kg m^{-2} for single mouldboard ploughing operations. The particle size correction factor could not be measured directly, and a constant value of 1.00 was set. Additionally required transect information for the mass balance model 3, namely segment length and slope gradients for individual hillslope sections, were derived from the digital elevation models of the hillslopes.

Table 3.3 Input parameters for the conversion models

Model parameters	Dano	Wahable	Conversion model
Fallout input filename	north. fall	north. fall	MB-2; MB-3
Reference inventory Aref (Bq m^{-2}):	729	729	PM; MB-1; MB-2; MB-3
Year of sample collection t:	2005	2006	PM; MB-1; MB-2; MB-3
Bulk density of soil B (kg m^{-2}):	1150	1410	PM; MB-1
Plough depth d (m):	0.1	0.1	PM; MB-1
Start of cultivation to:	1954	1954	MB-2; MB-3
Proportion factor y:	0.5	0.5	MB-2; MB-3
Relaxation depth H (kg m^{-2})	4.00	4.00	MB-2; MB-3
Plough depth d (kg m^{-2})	115	141	MB-2; MB-3
Tillage constant Φ (kg yr-1 m^{-1})	200	250	MB-3

PM: Proportional model; MB-1: Mass-balance model 1; MB-2: Mass-balance model 2; MB-3: Mass-balance model 3

3.4 Modeling approach

3.4.1 Description of the Water Erosion Prediction Project (WEPP) model

Soil erosion at hillslope scale was simulated by the Water Erosion Prediction Project (WEPP) model (version 2006.5). The WEPP model, developed by Flanagan and Nearing (1995), is a physically-based model which simulates spatially distributed temporally dynamic soil erosion and runoff processes. The hillslope version considers two-dimensional erosion processes along the slope profiles and is therefore limited to the simulation of runoff and soil erosion in simple and morphologically not very complex landscapes, which was given for both hillslopes. The continuous simulation mode was used to predict average annual runoff and soil loss over a time interval of 50 years. Additionally, soil conservation options were integrated in simulation scenarios to reflect land management conditions at the study sites.

3.4.2 Input parameterization

The WEPP model requires a minimum of four input data files for simulations: a climate file, a slope file, a soil file and a land management file. Parameter information for all input files are either based on field investigations in Burkina Faso or derived from secondary data sources, such as expert knowledge and scientific literature.

Climate

For continuous climate simulations, the stochastic daily weather generator CLIGEN (Ver 4.3) was used, which is a stand-alone program of the WEPP model (Nicks et al., 1995) requiring averaged monthly weather data to generate a representative weather pattern for long-term time intervals. Based on continuous 8-years climate data for the study site, CLIGEN was used to generate synthetically weather data for a time-period of 50 years.

Daily rainfall data from 1999-2005 were obtained from the Regional Agricultural and Hydrological service of the Ioba province (DPAHRH, Burkina Faso)[9], which provided reliable rain-gauged data for Dano. Continuous daily data on temperature, solar radiation and rainfall intensities (10-min. intervals) were available for the year 2006 from a climate station in Dano installed by the Glowa-Volta Project (ZEF, University of Bonn). As the variability of temperature and solar radiation between different years is low, its effect on runoff and soil loss simulations is expected to be minor. Firstly, temperature and solar radiation in these tropical conditions will not descend below or exceed the minimum or maximum required values for plant growth (e.g., millet), secondly, even if a temperature increase within the last 50 years occurs, average temperature variations of 1-2 °C will still have a small effect on evapotranspiration and hence on soil-water balance and vegetation cover. Finally, temperature correlates within CLIGEN only in terms of higher, respectively, lower temperature values depending on the probability of rainfall, which means that temperatures tend to be higher on dry days following a dry day and lower on wet days following a wet day (Nicks et al., 1995). Therefore, the one-year temperature and solar radiation data were assumed to be representative also for longer time periods.

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[9] Direction régionale de l'Agriculture de l'Hydraulique et des Ressources Halieutiques du Sud-Ouest, District du Ioba, Diébougou, Burkina Faso

The probabilities of precipitation events as well as the mean, standard deviation and skewness coefficients of daily precipitation of each month were directly extracted from rainfall records in Dano. The first-order Markov model was used to generate the sequence of wet days following a wet day and dry days following a dry day within each year (Elliot and Arnold, 2000; Nicks et al., 1995). The normal distribution pattern of precipitation amounts and its cumulative probability within a year shows that approximately 50 % of all precipitation events have rainfall amounts higher than 10mm, 30 % have rainfall amounts higher than 20mm, 10 % higher than 30mm and about 3 % of all rainfall events will cause amounts higher than 50mm. Beside these relatively high amounts of rainfalls during single events, most of these highly erosive events show high rainfall intensities, which are characteristic for tropical conditions. Therefore also the storm pattern of each rainfall event was considered by calculating its effective duration, its maximum 30 min. intensity and its relative time to peak intensity for each month. Furthermore, an additional control dataset based on continuous rainfall data of 2006 was generated by the Breakpoint Climate Data Generator of WEPP (BPCDG) (Zeleke et al., 1999; Zeleke, 2001) in order to analyze the intensity, magnitude and frequency of single rainfalls events within a specific year.

The effect of wind velocity and direction is considered within CLIGEN only in relation to snow accumulation, snow melting and evapotranspiration, not in its function as active medium for soil particle detachment, transport and deposition (Nicks et al., 1995). As the sensitivity of the model to any changes concerning wind parameters is minor and continuous wind data were not available, default parameters were used.

Slope

Topographical data for the slope file was derived from cross sections of the DEM (Figure 3.2 and 3.3) using Surfer software. The input file requires information about slope gradients (%), slope length, slope shape and slope aspect. The curve view option of the WEPP model was used and input parameters were provided for single points at a defined distance along the hillslope profile. An equal distance of approximately 15-20 m spacing between single slope points were chosen resulting in an accumulated distance of 350 m for the slope line in Dano and 240 m for that in Wahable. For model simulations, cross section 2 of each site was selected as most representative concerning slope and soil characteristics.

Soil

Input data for the soil file were provided by field investigation of six soil profiles along a representative transect in Dano and five soil profiles along a representative transect in Wahable. Soil horizon properties for each soil layer include information about soil texture, organic matter, potential cation exchange capacity (CEC_{pot}), rock fragments and bulk density, which were available from field measurements.

A sensitivity analysis has shown that the model responds mainly to soil parameter changes in the upper soil layer (0-20 cm), which is most decisive for models` internally calculated soil-water balance and its hydraulic processes. While slight changes in runoff and soil loss occurred due to soil parameter changes in the second layer (20-40 cm), they were approaching zero in the third layer (40-60 cm) and not observable beyond a depth of 60 cm (Brunner et al., 2004). Therefore, soil parameters for the first two soil horizons were integrated in the WEPP model and the accumulative depth of each soil layer - as it was described in the field - was internally adjusted to a new set of soil layers with a standardized depth for model runs. Additionally, the sensitivity analysis indicated that simulations including more soil information by a larger number of individual soil profiles may decrease the uncertainty of spatial soil variability and may generate a more accurate presentation of soil erosion and deposition zones at a hillslope (Brunner et al., 2004). Therefore, all individual soil profiles were considered as slope specific soil units in simulations and soil profile parameters for transect 2 both in Dano and Wahable were chosen as most common soil characteristics for the specific hillslopes conditions.

Required hydraulic characteristics and soil erodibility parameters for each profile, such as effective hydraulic conductivity, baseline rill and interrill erodibility and critical shear stress were internally calculated by the WEPP model based on pedo-transfer functions (Flanagan and Livingstone, 1995). Soil surface albedo was calculated to be 17.4 %, based on average monthly solar radiation values provided by a climate station in Dano for 2006. The initial saturation level was estimated to be very low (0-1 %) as the first day of the first simulation run (1 January) is in the mid of the dry season without any rainfall for the last two month. However, as the continuous simulation mode was used for multiple simulation years, most of these parameters are automatically adjusted internally by the WEPP model.

Land management

The land management file consists of four specific input databases: an initial condition database reflecting vegetation conditions for the first simulation, a tillage database considering the effect of tillage by handhoe for Dano and ox-plough tillage for Wahable, a plant/harvest database including all required plant growth and crop yield parameters for sorghum (*Sorghum bicolor*) and an additional operation database reflecting cultivation practices, such as residue management, burning, contouring and construction of stone lines as feasible land management options for farmers. One rain-fed cultivation period was simulated annually determined by the uni-modal rainfall pattern from May to October.

For the initial condition database, which starts on the first January of the first simulation year, a ground coverage of 25 % vegetation was estimated to reflect vegetation cover conditions in the mid of the dry season, where many fields are either burned after harvest and/or the remaining ground cover is withered due to the dry climate conditions. While these initial conditions might have a significant effect on short-term and event-based simulations, its effect on long-term simulations is minor as its values are continuously internally adjusted based on previous growing seasons.

In terms of tillage, farmers prepare their soils only once a year mainly by handhoe (Dano) and - more rarely - by ox-plough (Wahable). For handhoe practices, tillage and planting are assumed to be on the same date – 30 June – whereas for ox-plough practices the tillage date is assumed to be two weeks before planting – 15 June. Field preparation by handhoe in which small planting holes are dug can be seen as a conservative tillage method due to its low surface disturbance. Tillage by ox-plough will cause a higher surface disturbance and might create temporarily a ridge-furrow system, which will have both an impact on soil compaction and an effect on models' calculation of rill and interrill areas. However, the ox-plough tillage in Wahable is performed along contour lines, which will decrease runoff and reduce sediment transport during intensive rain storms.

The plant/harvest database was generated for sorghum (*Sorghum bicolor*), because this starch plant is known as an important food crop for humans and animals in Sub-Saharan Africa (Cirad et al., 2002) and is also the dominant agricultural plant at both study sites. Crop specific information about production, eco-physiology, cultivation, harvest and yield could be obtained from scientific literature for West Africa (Cirad et al., 2002; Rehm and Espig, 1991) and was included in the available WEPP database (Arnold et al., 1995; Stott et al., 2005).

The effect of actual land management practices, such as burning after harvest, contour farming and construction of stone lines and possible future land management options, such s residue management and minimum tillage was evaluated and its benefits and limitations discussed. The effect of contouring was simulated for ox-plough tillage along contour lines - as it is performed in Wahable - by considering a minimum ridge height of 0.1 m and a spacing of 0.8 m between rows. The integration of a permanent ridge-furrow system is seen as a way to capture the effect of stone lines along contours in a simplified manner. Therefore, in simulation scenarios for Dano, stone rows were accounted for by assuming a minimum ridge height of 0.3 m, a maximum side slope of 3° along the rows and a spacing of 5 m, 10 m and 15 m between rows.

Burning is seen as a common practice on farmers field immediately after harvest and has a significant impact on the vegetation cover and biomass production. Within the model this practice was included by considering both the loss of flat residue biomass (50 % and 75 %) and the loss of standing residue biomass (50 % and 75 %). Another wide-spread application is to leave stalks on the field or to add plant residue as complementary soil cover and supplementary biomass source. For simulation scenarios, added residue after harvest was assumed to be 0.3 kg m^{-2}, which might be a desirable although not always feasible amount of residue given the limited biomass resources. Additionally, the influence of minimum versus high-input tillage on runoff and sediment delivery was compared as an example of conservation versus conventional tillage. For minimum tillage, all parameters of the tillage database were set zero, except the required but very low-set values for random roughness after tillage (1.5), surface area disturbance (15 %) and tillage depth for planting holes (5 cm). For conventional tillage, an optional operation database of the WEPP model was used (“chisel plough”), which causes a high surface disturbance (100 %) and has a mean tillage depth of 15 cm.

3.4.3 Delineation of overland flow elements

The WEPP model offers the option to divide the hillslope into a maximum of 10 individual hillslope units, which are called Overland Flow Elements (OFE). Each OFE presents a specific factor combination of climate, slope, soil and land management (Flanagan and Livingston, 1995). In this study, the delineation of the hillslopes was based on changes in both soil and slope conditions assuming all parameters for climate and land management to be uniform along the slopes. Terrain attributes were used to delineate the hillslopes into five slope positions, namely summit, shoulder, backslope, footslope and toeslope/valley position. Soil characteristics were assumed to be deterministic and, therefore, representative for individual hillslope positions. According to the number of soil profiles and changes in slope gradient steepness, 6 OFEs were differentiated for the hillslope in Dano and 5 OFEs for the hillslope in Wahable. The length of each section varied depending on soil and slope characteristics and was adjusted to the available field information from soil profile investigations along the hillslope. An earlier study has already shown that a hillslope differentiation into five OFEs can account sufficiently for the given variability of tropical soils and can produce a clear distinction between soil erosion and accumulation zones in modeling approaches by WEPP model (Brunner et al., 2004).

Additionally, this delineation into several smaller hillslope units helps meet the requirements of the appropriate scale as the recommended slope length for hillslope simulations is tens of meters with a maximum length of 100m (Ascough et al., 1997).

3.5 Results and discussion

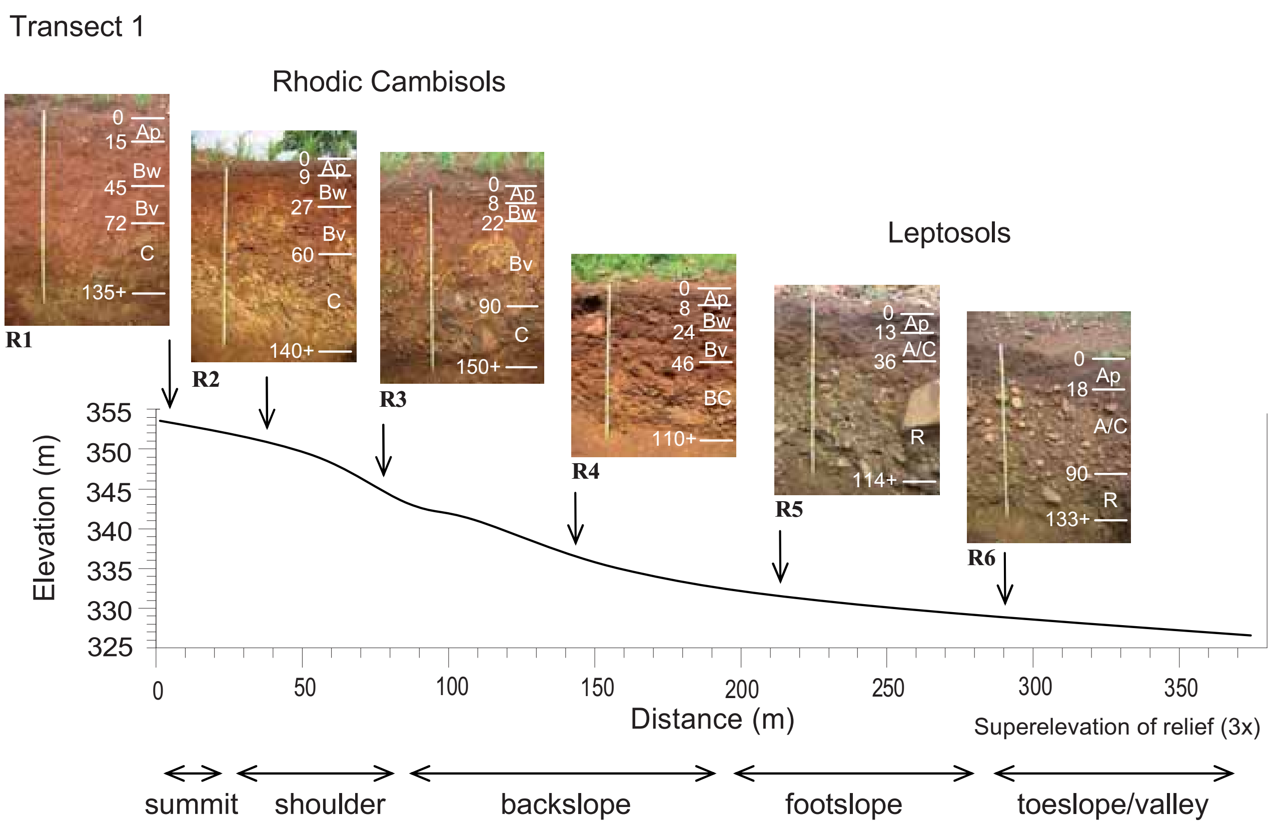
3.5.1 Soil catenary development

The descriptions of the soil profiles indicate that diagnostic soil properties are closely related to the pedo-geomorphic processes ascribed to each individual hillslope position. The dynamic interaction between soil formation and landform characteristics confirms furthermore the concept of catenary soil development as it was first developed by Milne (1935) for soils in East Africa and further specified by Conacher and Dalrymple (1977) in the nine-unit landsurface model. The nine-unit landsurface model is a soil-water-gravity model explaining the differences in soil properties by their geomorphologic position within the landscape, which influence in return the mobilizations, translocation and redeposition of soil material. Based on this landsurface model, the diagnostic soil properties along two transects at each study site are described and interpreted with regard to their pedo-geomorphologic features.

Along the catena in Dano (Figure 3.4a and b), soils at the upper hillslope can be classified as *rhodic cambisols* (WRB, 1998), which correspond with *sol brun eutrophe peu évolué faciès ferruginisé* (CPCS, 1967). These soils have a fine clay to silty clay texture, pH-value of 6, high cation exchange capacity, and moderately to highly developed structure. The existing, but weakly developed, illuvation horizon (Bw) reflects an accumulation of clay in the B-Horizon. This formation of argillic subsurface horizons is a typical phenomenon and often reported for upper slope sections (Stoop, 1987). Although the four upper soil profiles show similar soil characteristics and are assigned to the same soil group of cambisols, their individual diagnostic properties vary depending on their topographic position. For example, for both transects, the soil profiles at the relatively flat summit position have a thick A-horizon of 15 cm (R1) and 27 cm (L1), which diminishes downslope to 9 cm (R2) and 10 cm (L2) at the shoulder position and to 8 cm (R3, R4 and L3, L4) at the backslope position. This might be due to the dynamic interaction between soil properties and slope geometry. Whereas the slope at the summit position (or interfluve) is relatively flat, and pedogenetic processes are dominated by vertical movement of subsurface soil water, the slightly convex slope at the shoulder position leads to increased soil creep. However, the summit position is not as pronounced as in a classical example because the hilltop is narrow (< 30 m in each direction) and the slope gradients are higher than 5°. A transition zone between summit and shoulder position has probably already been reached. The convexity at the shoulder position indicates high intensities of soil erosion, and slope gradients of 25° present maximum values for the hillslope. Pedo-geomorphic processes include the lateral movement of subsurface soil water and seepage and, associated with this, the occurrence of high concentrations of manganese and iron oxide leading to the formation of subsurface concretions at soil depths of 60 cm (R2) and 65 cm (L2).

The backslope position shows an almost straight geometry, which is associated with the transport of large amounts of soil material downslope. This zone is characterized by water-transmitting processes and surface wash. The slight irregularity in slope geometry (small rise at a distance of 100 m) is probably caused by stone lines constructed along elevation contours, which lead to a flattening and also to a slight elevation in microtopography. In contrast to other parts of the hillslope where remnants of soil conservation techniques can also be observed, stone lines at the lower shoulder and upper backslope positions are in a good condition, relatively high (approximately 15-20 cm) and in small regular intervals. The effect is a flattening of the slope similar to that of terraces, which leads to a reduction in soil movement downslope.

Soils at the lower hillslope positions, such as footslope and toeslope, differ significantly from upper soils. On transect 1, *leptosols* (WRB, 1998), which are equivalent to *sol peu évolué d'apport colluvial modal* (CPCS, 1967), are found. These degraded soils are characterized by a succession of A/AC and C horizons and show a very high content of coarse rock fragments or/and a continuous hard cover near the surface (R5 and R6). The limited water-holding capacity and the skeletic subsoil layer restrict their potential for agriculture. Leptosols present an initial stage of soil formation and are generally located on steep slopes, namely shoulder or backslope position, where soil formation is restricted by erosion. Here, leptosols were found at footslope and toeslope positions, where usually deeper depositional soils are expected. The apparent contradiction in the catenary sequence might be due to former soil erosion at these positions and a more recently begun soil formation process. Another assumption could also be that the skeletic AC layer is a buried horizon resulting from ancient redepositional processes where debris material from upslope was transported downhill. Heterogeneous, unsorted colluvial deposit, often also characterised by discontinuous and disturbed paleosol horizons, are frequently found at footslope/toeslope positions.



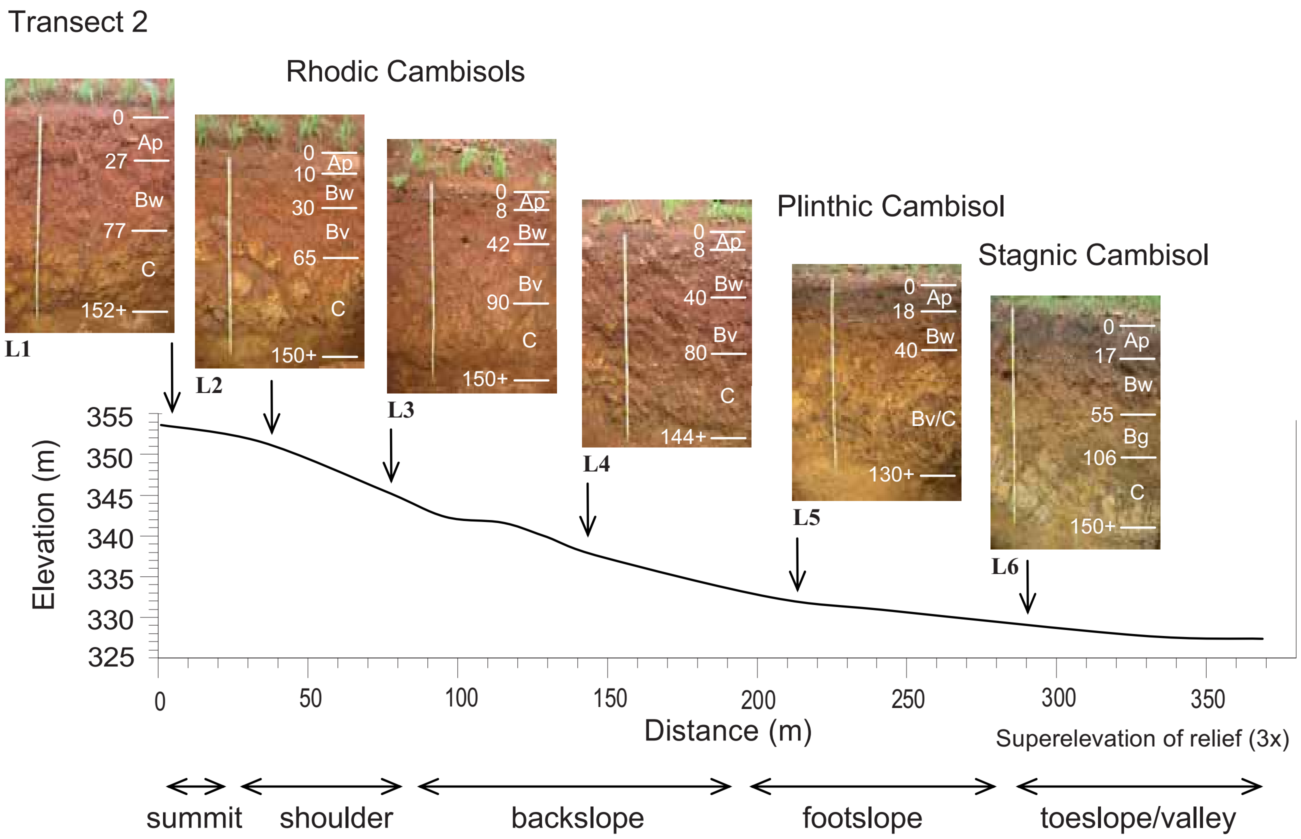
Abbreviations for master and transitional horizons (Schoeneberger et al., 2002)

A: Mineral soil, formed at surface; accumulation of humified organic matter but dominated by mineral matter; cultivation properties. **A/B or A/C:** Discrete, intermingled bodies of two horizons: A and B or C material, majority of layer is A horizon material. **B:** Mineral soil, typically formed below O,A, or E (typical features e.g. illuvial accumulation, gleying). **B/C:** Dominantly B horizon characteristics but also has some recognizable characteristics of the C horizon. **C:** Mineral soil, soft bedrock; layer little affected by pedogenesis; may or may not be related to parent material of the solum. **R:** Hard bedrock (continuous, coherent strongly to indurated cemented.

p: Tillage or other disturbance of surface layer (e.g. pasture, plow).

v: Plinthite (high Fe, low OM, reddish contents; firm to very firm moist consistence; irreversible hardening with repeated wetting and drying). **w:** Minimal illuviation accumulation.

Figure 3.4a Soil catena for cross sections (transect 1) at Dano hillslope, Burkina Faso



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g: Strong gley, typically ≤ 2 chroma; may have other redoximorphic (RMF) features. **p:** Tillage or other disturbance of surface layer (e.g. pasture, plow). **v:** Plinthite (high Fe, low OM, reddish contents; firm to very firm moist consistence; irreversible hardening with repeated wetting and drying). **w:** Minimal illuviation accumulation.

Figure 3.4b Soil catena for cross sections (transect 2) at Dano hillslope, Burkina Faso

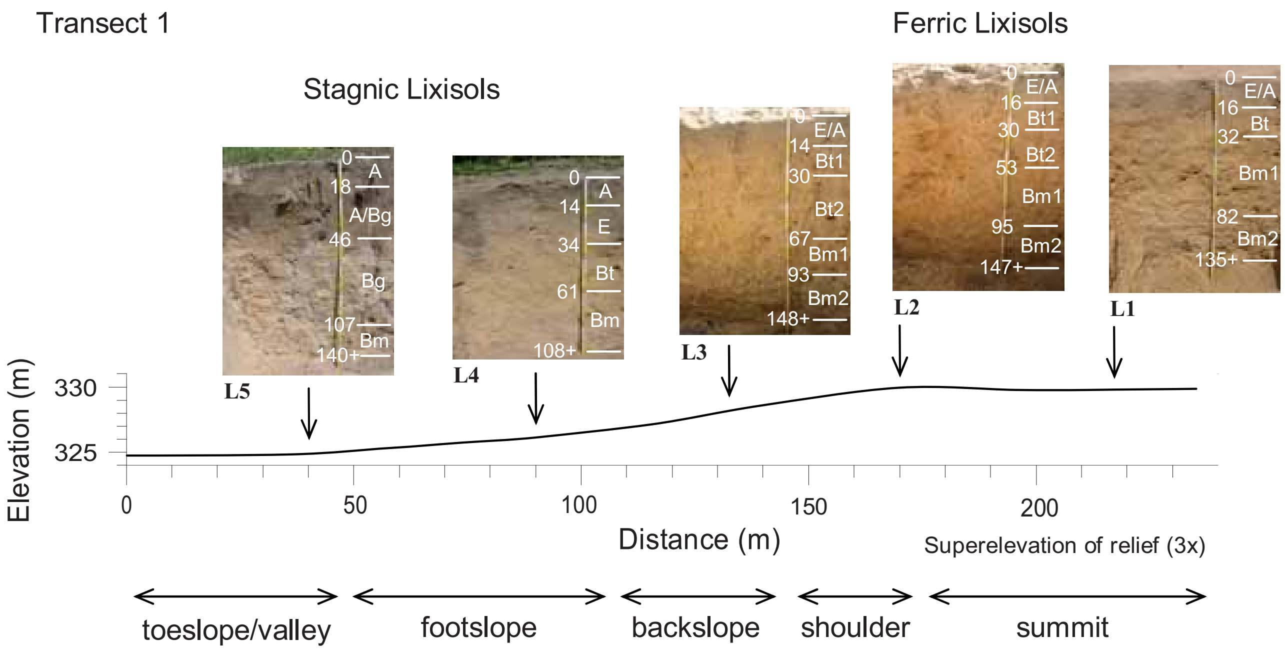


Figure 3.5a Soil catena for cross sections (transect 1) at Wahable hillslope, Burkina Faso

Abbreviations for master and transitional horizons (Schoeneberger et al., 2002)

A: Mineral soil, formed at surface; accumulation of humified organic matter but dominated by mineral matter; cultivation properties. **A/B:** Discrete, intermingled bodies of two horizons: A and B or C material, majority of layer is A horizon material. **B:** Mineral soil, typically formed below O,A, or E (typical features e.g. illuvial accumulation, gleying). **E:** Mineral soil, loss of clay, iron, aluminum and/or organic matter leaving a net concentration of sand and silt; lighter color and coarser texture. **E/A:** Discrete, intermingled bodies of two horizons: E and A material, majority of layer is E horizon material.
g: Strong gley, typically ≤ 2 chroma; may have other redoximorphic (RMF) features. **m:** Strong pedogenic cementation or induration (>90% cemented, even if fractured); physically root restrictive. **t:** Illuvial accumulation of silicate clays (clayskins, lamelae or clay bridging).

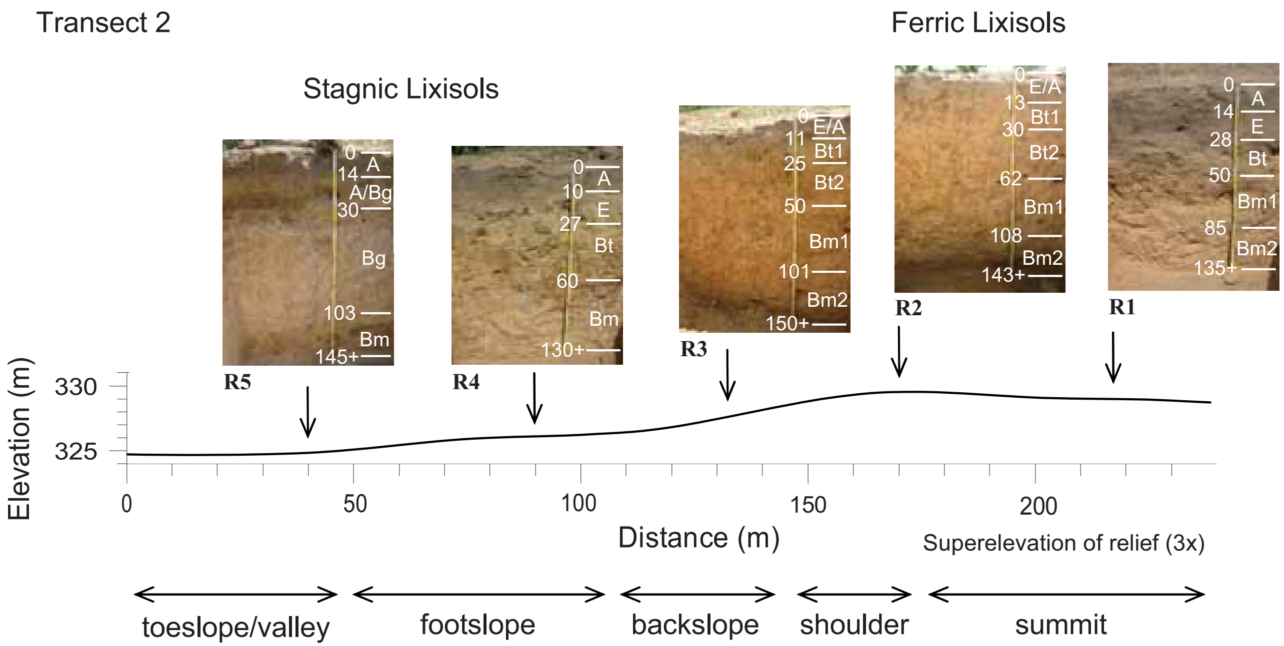


Figure 3.5b Soil catena for cross sections (transect 2) at Wahable hillslope, Burkina Faso

Abbreviations for master and transitional horizons (Schoeneberger et al., 2002)

A: Mineral soil, formed at surface; accumulation of humified organic matter but dominated by mineral matter; cultivation properties. **A/B:** Discrete, intermingled bodies of two horizons: A and B or C material, majority of layer is A horizon material. **B:** Mineral soil, typically formed below O,A, or E (typical features e.g. illuvial accumulation, gleying). **E:** Mineral soil, loss of clay, iron, aluminum and/or organic matter leaving a net concentration of sand and silt; lighter color and coarser texture. **E/A:** Discrete, intermingled bodies of two horizons: E and A material, majority of layer is E horizon material.
g: Strong gley, typically ≤ 2 chroma; may have other redoximorphic (RMF) features. **m:** Strong pedogenic cementation or induration (>90% cemented, even if fractured); physically root restrictive. **t:** Illuvial accumulation of silicate clays (clayskins, lamelae or clay bridging).

Soils at the same positions on transect 2 are classified as *plinthic cambisols* (WRB, 1998) and *stagnic cambisols* (WRB, 1998), which correspond to *sol peu évolué d'érosion lithique* and *sol brun eutrophe peu évolué hydromorphique* (CPCS, 1967), respectively. These soils are either restricted by a plinthic hard-rock layer at a shallow depth of 30 cm (L5) or show hydromorphic features and eluvial horizons (L6), which limit their potential for agriculture. Nevertheless, the good physical and chemical properties of the A-horizon, namely its fine clay loam to loam texture, pH value of 6, high cation exchange capacity, and richness in exchangeable bases still allows the cultivation of annual crops, such as millet (*Pennisetum glaucum*) and sorghum (*Sorghum bicolor*).

Irrespective of their different diagnostic horizons and soil classification, soils at the footslope show similar characteristics due to their hillslope position. At both transects, the slope geometry is almost straight and only slightly concave; slope gradients of 5-10° are significantly lower than the 10-20° of the backslope position. Colluvial redeposition processes from upslope predominate, and the smooth concavity refers to surface wash and permanent cultivation activities. The A-horizon thickness increases continuously in comparison with upper soil profiles from 8 cm at the backslope position (R4, L4) to 17-18 cm at the toeslope positon (R6 and L6). Furthermore, at the toeslope position, water retention characteristics, such as mottles and iron oxides are more distinct than at the former position. There is no explicit valley position, as the landscape continues slightly undulating with slope gradients of 1-3° extending over approximately 1-2 km until a final - or in geological terms intermediate – sink, the dam, is reached.

In Wahable, soils at the upper hillslope are *ferric lixisols* (WRB, 1998), which correspond with *sol ferrugineux tropicaux lessivé* (CPCS, 1967) (Figure 3.5a and b). These soils have a shallow ochric A-horizon followed by a chemically depleted eluvial horizon (E or E/A). Below the eluvial horizon, a moderate accumulation of low activity clays has lead to the formation of an argic horizon (Bt) dominated by kaolinitic clays. The high content of iron and manganese oxides gives the soil its typical reddish or ochre color. They have a sandy loam to sandy clay loam texture, medium soil depth, and low organic matter content. As they are chemically depleted and/or leached, the potential soil fertility is generally very low. A massive, cemented Bm horizon is visible in all lower parts of the soil profiles, generally below a depth of 50 cm (except L1), which might indicate a transition into plinthic horizons, typical for tropical and subtropical regions.

The slightly inclined, almost straight slope geometry has low average slope gradients of 2-3°, which reach a maximum of 10° at steepest shoulder/backslope positions and decrease to less than 1° at footslope/toeslope positions. Despite the low relief energy, a high amount of runoff with very high flow velocities can be generated during heavy rainstorms, mainly due to limited infiltration capacities and high surface sealing of the soil.

The summit position is flat, and soils are mainly influenced by vertical soil formation processes. However, the A-horizon at the summit and shoulder position (L1, L2 and R1, R2) is only 2-3 cm thicker than at the backslope position (L3, R3), and permanent and intensive cultivation have already led to diminished soil fertility. The slight slope convexity at the shoulder position indicates lateral soil/water movement, soil creep and chemical eluviation of subsurface soil. The backslope position is characterized by a straight slope referring to an equilibrium between influx from upslope and outflow downslope (Carson and Kirkby, 1972).

On lower footslope positions, *stagnic lixisols* (WRB, 1998), which are equivalent to *sol ferrugineux tropicaux lessivé hydromorphe* (CPCS, 1967), characterized by alternating reducing and oxidizing conditions are found. These soils are temporarily completely saturated with surface water from upslope. Due to the good saturated conditions, the soil is often used to cultivate rice (*Oryza sativa*). Soils have a loam to sandy loam texture, a greyish colour and reddish brown (ferrihydrite) and yellowish brown (goethite) mottles as well as iron and manganese oxides in the subsurface layer. The intensity of hydromophic features increases from the higher footslope position (L4, R4) toward the lower toeslope/valley position (L5, R5) due to a more frequent saturation by groundwater. At transect 1, the slope geometry is straight to slightly concave indicating surface wash processes. The straight to slightly convex slope geometry at transect 2 might be attributed to a lowering of the base level and beginning regressive erosion. However, at both transects, the increasing A-horizon thickness especially at the toeslope and valley position with 18 cm (L5) and 14 cm (R5) respectively, clearly indicate soil movement and colluvial redeposition from upslope.

Even though there is no distinct valley position here either, a natural channel appears immediately behind the last soil profile and functions as a discharge, transporting water and soil material into the dam of Wahable, approximately 500 m away.

The soil profiles at both study sites show that an interrelation exists between slope geometry and variations in physical and hydrological soil properties. Furthermore, it can be concluded that the concept of catenary soil development described by Conacher and Dalrymple (1977) is confirmed for the two soil transects at each study site, where the systematic sequence of soils can be interpreted as a result of specific morphological and physical conditions downslope.

The soil-landscape description of the two catenae in Dano and Wahable illustrates that these landsurface units - here referred to as hillslope positions - show a typical slope geometry and inherent diagnostic soil characteristics, such as eluviated horizons, leaching, water retention and hydromorphic features. However, it is not common to find all units on a single landform, and also here only five units of the nine-unit landsurface model could be identified clearly, namely summit (interfluve), shoulder (convex creep slope), backslope (transportational midslope), footslope (colluvial footslope) and toeslope (alluvial toeslope). It should also be emphasized that even though soil-water processes might be predominating on one specific hillslope unit, most processes operate also on other hillslope units, and only the "relative differences amongst their responses" (Conacher and Dalrymple, 1977, p. 101) lead to a clear identification.

Furthermore, a comparison between the different catenae in Dano and Wahable indicates that slope geometry might be more important than slope gradient. Although slope gradients at the hillslope in Dano are steeper, and the relief energy is four times higher than that of Wahable, a more distinct catena could be identified in Wahable. This might be explained by the geomorphological history of the landscape, the geological formation and long-time weathering processes. While slope geometry has an impact on the spatial redistribution of soil and water, slope gradients and intensities might determine the duration of temporal processes and the period of geomorphologic changes in landform. Mulders et al. (2001) found the same main soil types along a toposequence in the central-northern region of Burkina Faso (Sanmatenga), where he identified relatively fertile cambisols in a hilly Birrimian area characterized by high local relief and unfertile sandy loam lixisoils in Ante-Birrimian areas with slightly undulating, low local relief. This corresponds very well with the findings of this research, where clayey cambisols were found on the topographically relatively pronounced hillslope in Dano and sandy lixisols on the almost flat area of Wahable.

Burrough (1993) emphasized that "the estimation of the variance of a soil property has little meaning unless it is expressed in regard to the size and kind of spatial units for which it was estimated". Therefore, catenary soil development information is used to differentiate the hillslope into various soil-landscape or pedo-geomorphologic process-response units, which can be included later in environmental modeling approaches such as soil erosion simulations (see sections 3.57 and 3.58).

While for this study a discrete approach is used that divides the hillslope into distinct catena units based on descriptive, qualitative soil profile information, many other studies follow a continuous, deterministic or stochastic approach, which is based on mathematical functions such as regression analysis or geostatistics (Burrough, 1993; McBratney et al., 2000; Park and Vlek, 2002). Although these approaches are often preferred due to their mathematical/statistical nature, clearly defined sampling designs and hence supposed higher objectivity, they have the disadvantage that they require a very large sampling dataset to capture the spatial variability of soils. They also have several limitations in terms of parameterization procedures for models and extrapolation to other areas (Burrough, 1993; Hoosbeek and Bryant, 1992). Therefore, in recent years, discrete approaches have been re-investigated and seen as adequate both to identify spatial domains of similar process dynamics over hillslope or landscape scales and to derive input datasets for modeling procedures based on these discrete spatial domains (Blöschl and Sivapalan, 1995; Flügel et al., 2003; Märker, 2001). These descriptive, qualitative approaches might have a "comparative advantage over the continuous approach as it reduces the effort and time to collect necessary soil parameters over the landscape", but limitations remain. They suffer "from arbitrariness of criteria used to delineate the landscape and also from the inability to capture the internal variability of parameters within each spatial domain or unit" (Park and van de Giessen, 2004, p. 29 and p. 30, respectively). A promising new approach is, therefore, to combine the advantages of the continuous and discrete approach by first following the catenary concept, which allows identification of pedo-geomorphological units along a transect, and then reinterpreting this qualitative concept by using continuity equations. This allows simulations of water and material flows over the hillslope (Park et al., 2001; Park and van de Giessen, 2004). Park et al. (2001) developed a simple, process-based terrain characterization index, which presents a quantitative tool to explain the spatial variability of soil properties in interrelation with surface topography and is easily transferable to other areas. For future investigations, this terrain characterization index could be applied as a more advanced method for the delineation of hillslope units. Moreover, it could be used to facilitate and confirm the identification of spatial variations in soil properties and to derive input parameters more systematically for soil erosion modeling in West Africa.

3.5.2 Variability of soil properties along a catena

The soil profile investigations confirmed catenary soil development along the slopes. The analysis of chemical and physical soil parameters shows that variations in quantitative soil properties such as A-horizon thickness, bulk density, organic matter, potential cation exchange capacity (CEC_{pot}), soil available P ($P_{available}$) and soil available K ($K_{available}$), total nitrogen (N), carbon-nitrogen ratio (C/N-ratio) and soil texture are related to individual hillslope positions (Figure 3.6 and 3.7).

A comparison between both study sites indicates that most physical and chemical topsoil attributes show a similar trend downslope, although their absolute values might differ highly depending on the contrast in soil types and land management practices. At both sites, for example, the solumn thickness of the A-horizon decreases from summit to shoulder and backslope position and increases again at toeslope/valley position. At the same time, the percentage of organic matter, which is closely related to the A-horizon thickness, generally increases downslope (with the exception of the upper backslope at Dano and the footslope position at Wahable) and corresponds well with the redeposition of soil material from upslope. The percentage of organic matter is medium to high at Dano (average 2.2 %), but low to very low at Wahable (average 0.65 %), which indicates a low fertility status of the sandy soils. Organic matter content is a key element for soil fertility (Pieri, 1989; Feller, 1995) and has an essential influence on biological activities and soil physical properties (e.g., infiltration rates, crust formation).

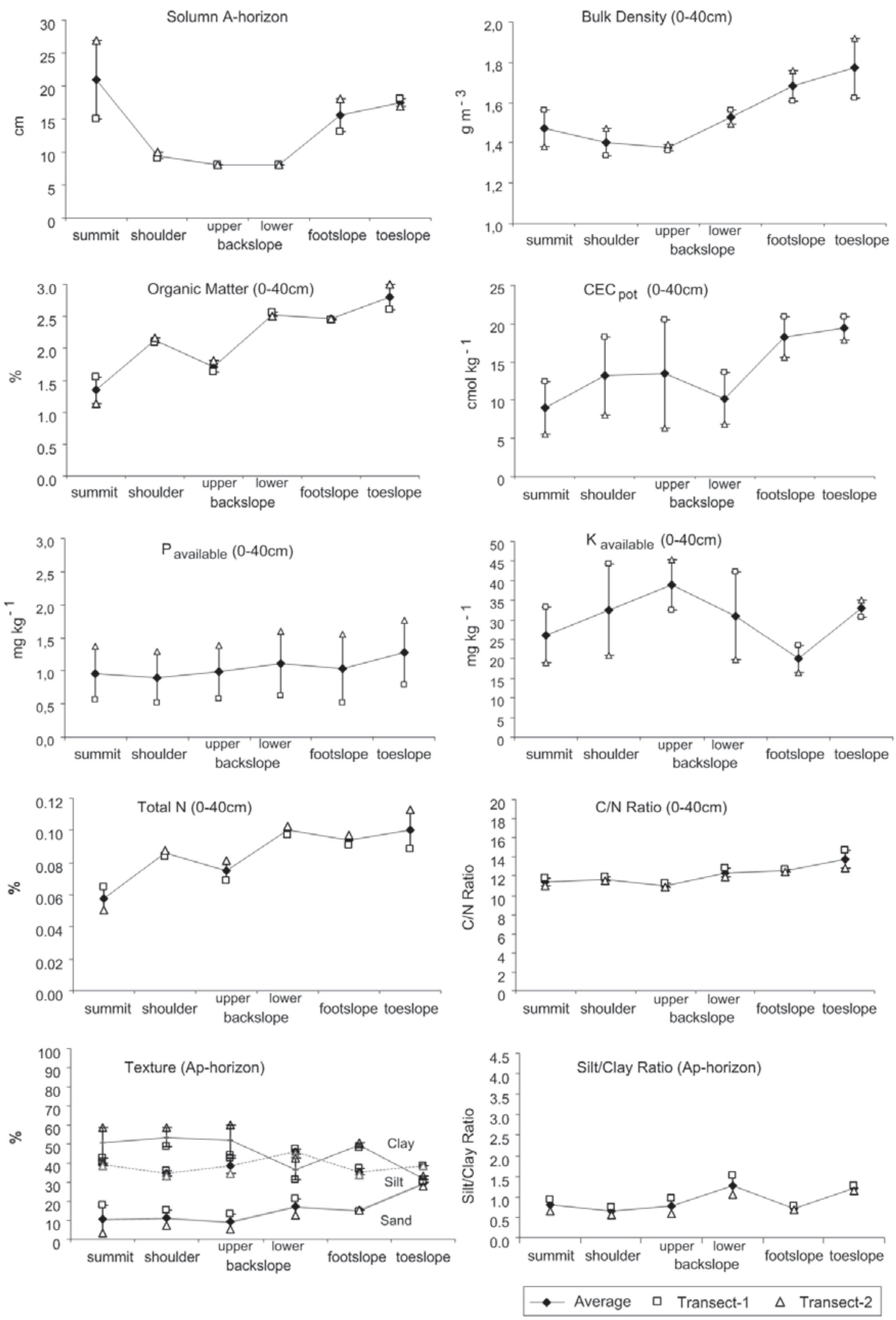


Figure 3.6 Spatial variability of soil parameters along the catena at Dano, Burkina Faso

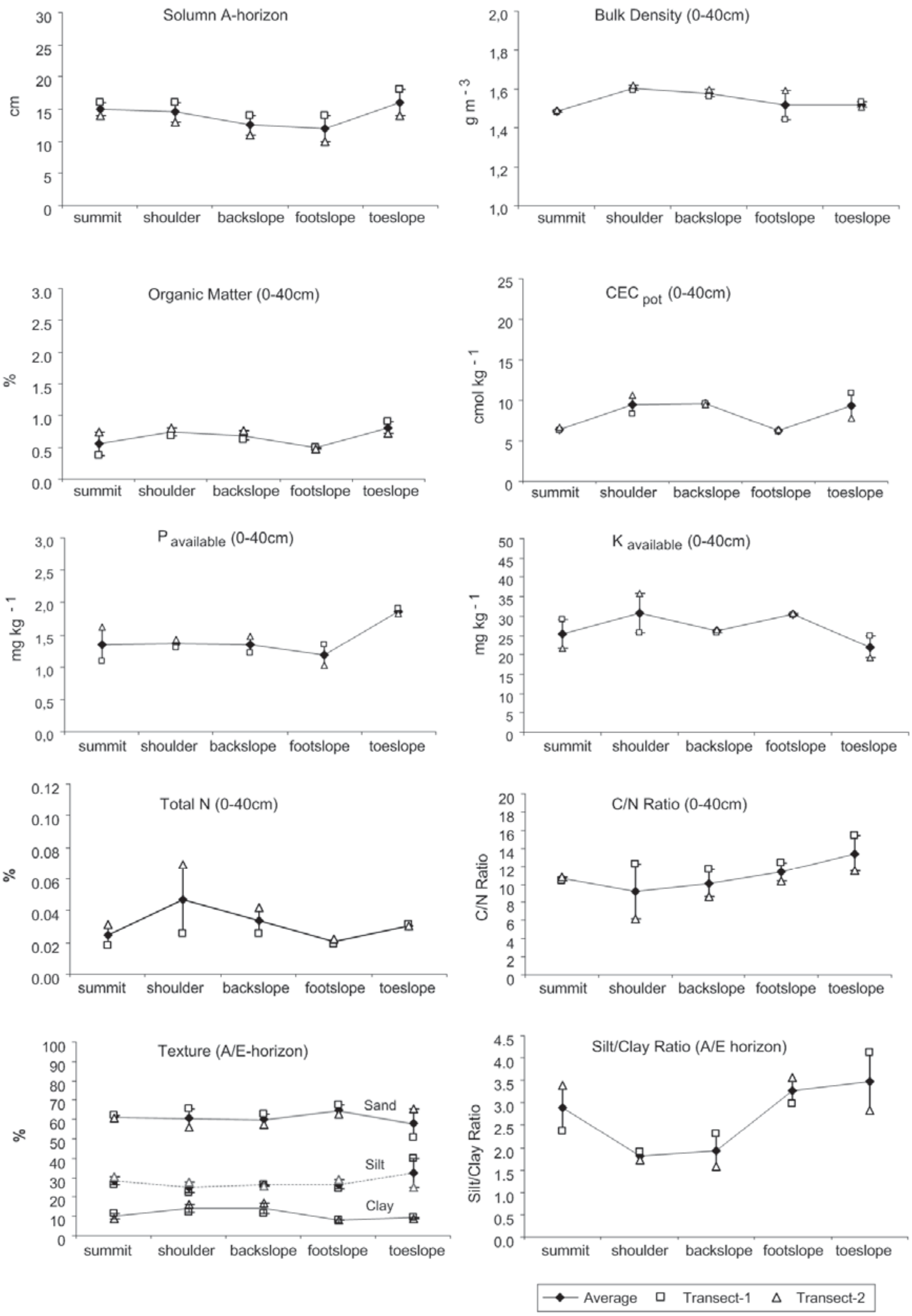


Figure 3.7 Spatial variability of soil parameters along the catena at Wahable, Burkina Faso

A decline in organic matter will lead to nutrient depletion and later to physical soil degradation (Floret and Pontanier, 1993; Kiepe et al., 2001a; Mando et al., 1999). Ouattara et al. (2006) also analyzed the influence of soil texture classes on soil organic matter content and found that soil organic matter increases with increasing clay content resulting in lower depletion rates. By comparing clay-rich ferric Luvisols and sandy ferric lixisols in Burkina Faso, he concluded that soil organic matter is generally more labile in the coarser sand fraction than in the silt or clay fractions of the soil, which corresponds well to the results for the soils at Wahable. Stoop (1987) analyzed soil properties along three toposequences in Burkina Faso and obtained very similar values for organic matter and reported the same increasing trend downslope.

Bulk density has very similar values at both sites (each average 1.54 g m^{-3}), but the downslope pattern and the reasons for the relatively high values are different. At Dano, bulk density decreases at shoulder and upper backslope position to 1.4 g m^{-3} and increases to above 1.7 g m^{-3} at the footslope and toeslope positions. The values are almost stable at Wahable except for a slight increase at the shoulder backslope position to 1.6 g m^{-3}. The relatively high bulk density values for the clayey soils at Dano might be caused by the high content of coarse material especially in lower hillslope positions (average percentage of coarse fragments 30 %, max. 45 % at toeslope positions). The relatively high bulk density values at Wahable might be due to soil compaction, which is further aggravated by land management practices such as ox-plough tillage.

Potential CEC is closely related to organic matter content and hence its pattern shows the same trend downslope. While the average CEC_{pot} is medium to high for clay soils at Dano (average 14 cmol kg^{-1}, max. 21 cmol kg^{-1}), it value is slightly above the critical limit of 5 cmol kg^{-1} for optimal plant growth in Wahable. The low values at Wahable can be a result of the highly weathered sandy soils with pH values around 4.6 (in contrast to 6.0 at Dano) and a clay fraction of only 10 %, mainly dominated by kaolinites. The spatial variability is low at Wahable, but very high at Dano; differences between transect 1 and transect 2 show an average standard deviation (SD) of 3.8 cmol kg^{-1} and an average coefficient of variation (CV) of 31 %. For the upper backslope positions even a SD of 7 cmol kg^{-1} and a CV of 52 % were found.

These high variations between the transects at Dano are also visible in terms of available phosphorus and available potassium. Here, average SD of 0.4 mg kg^{-1} for

$P_{available}$ and 7 mg kg^{-1} for $K_{available}$ are found, which would correspondent to CVs of 4 % and 23 %, respectively. Spatial variations are lower at Wahable, where average SDs are 0.1 mg kg^{-1} and 2.5 mg kg^{-1} and average CVs are 10 % and 9.5 % for $P_{available}$ and $K_{available}$, respectively. The main reason for the higher variations at Dano could be different land management and fertilizer practices.

In Wahable, the main area of the hillslope belongs to one farmer and only fields at the toeslope position are cultivated by another farmer. In contrast, the hillslope area of Dano is divided both horizontally and vertically into many different smaller fields cultivated by different farmers. The patchy structure and the individual, often unpredictable land management practices, such as fallow rotation, intercropping, straw residues left on the fields or burning after harvest might be the reason for the high variations in soil nutrient contents. Although fertilizers are rarely used on these sorghum-dominated rainfed fields, which are located at some distance from the homesteads (so-called bushfarms), applications may have been made in the past in some areas. Human activities have a distinct effect on the natural variability of soil properties through agricultural management and will eventually lead to a "superimposed variability" (Stoop, 1987, p. 244). Also, biological activities (e.g., termites) might cause changes in soil micro-structure and relocate soil particles and hence soil nutrients within the soil column. However, in order to interpret soil fertility and nutrient stocks more precisely, detailed insight into land history, land management such as crop rotation, fallow cycles, intercropping, fertilizer use and biological activities would be needed. This was beyond the scope of this study.

Nevertheless, a first overview over the nutrient status of the soils is provided. In general, soils at both sites are very poor in primary nutrients and show soil available P and K values that are not sufficient for optimal plant growth. The extremely low P concentrations reflect low inherent mineral P in these intensively weathered soils in which ferralitic iron-oxide concretions have been formed in the subsoil. These serve as considerable sinks for P (Frossard et al., 1995). Additionally, P is continuously taken up by the plants and lost through crop harvests, usually without being adequately compensated for. The extremely low P availability at Wahable is a result of intensive weathering and long-time cultivation. Very similar P values ranging from 0.8 to 1.7 mg kg^{-1} were recorded by Mulders et al. (2001) on degraded soils in Burkina Faso the central-northern province, which were confirmed by a national soil survey for cambisols and ferric lixisols in the Ioba province of southwestern Burkina Faso (BUNASOL, 2000). Also, many other studies confirm a low nutrient status and P deficiencies under similar environments in Burkina Faso (Boyer, 1970; Hien, 1995; Kiepe et al., 2001b) and northern Ghana (Abeoko and Tiessen, 1998, Kpongor, 2007). Bielders et al. (2002), Sterk et al. (2004); Visser et al. (2007) also reported considerable nutrient losses through wind erosion in Burkina Faso and Niger, where especially high amounts of P and N were exported from fields.

Total nitrogen is closely dependent on organic matter content as its main source, and hence its pattern follows the same trend as C and increases downslope. While the soils at Dano show a sufficient amount of total N for plant growth (average 0.09 %), N is very limited at Wahable (average 0.03 %) as a result of a low organic matter. Zougmoré et al. (2004b) found similar N values under sorghum cropping systems in Burkina Faso and recorded losses through harvested sorghum, leaching and volatilization, but also through soil erosion, especially under intensive rainfall when high amounts of N (about 43 % of the total N outflow) were transported downslope by the fine sediment fraction. However, at both study sites, the average C/N ratio increased slightly downslope from 12 to 14. This can be interpreted as medium to high compared with normal decomposition rates.

The texture classes show that the silt content increased downslope at both sites, whereas the clay content decreased. This might be due to strong illuviation processes and strong adsorption capacity of clay minerals, which prevent particles from being detached by raindrops and transported downslope. The increase in silty material downslope is characteristic for footslope and toeslope positions (Stoop, 1987) and leads to an increasing silt/clay ratio. The relatively high percentage of sand and silt also provokes strong sealing processes, which could be observed especially at Wahable during intensive rainstorms.

In general, it can be concluded that chemical and physical soil properties follow the catena concept, although the pattern downslope is not always clearly pronounced. The spatial variability of soil properties between parallel transects is lower for the sandy soils in Wahable than for the clayey soils of Dano (with exception of the C/N ratio). This reflects the more homogeneous soils and land management practices at

Wahable as mentioned above. On the other hand, it is also possible that the severely degraded soils at Wahable have reached a low-level equilibrium in which variations in nutrients are unlikely. This was also postulated by Gray (1999) in a soil quality study for the same soil type in southwestern Burkina Faso.

The soil nutrient status at both sites is very low and highly deficient in soil available P. This might be due to a negative nutrient balance, which has often been reported for soils of Sub-Saharan Africa (Bationo et al., 1998 and 2007; Breman, 2001; Cobo et al., 2010; Smaling, 2006; Stoorvogel and Smaling, 1990) where more nutrients are taken up by plants and lost by harvest than are replenished by fertilizers or nutrient recycling such as straw mulch or compost. The use of mineral fertilizers is recommended in such environments and seen as indispensable to improve the soil fertility to ensure adequate crop yields (Buerkert, 2001; Vanlauwe and Giller, 2006; Vlek, 1990; Wopereis, 2006). Kpongor (2007) illustrated for soils in Northern Ghana that the use of inorganic fertilizer (N and P applications) can lead to significant yield increases in sorghum grain and is moreover agronomically as well as economically feasibly for farmers. Zougmoré et al. (2004c) pointed to the economic benefits of combining soil conservation techniques such as stone rows and grass strips with nutrient inputs such as compost-N and urea and found a maximum yield increase of 106 % when applying compost with stone rows. Also, Roose et al. (1999) showed that traditional soil management practices might enhance biomass production and hence increase soil fertility on degraded soils in northern Burkina Faso.

While the problem of soil nutrient deficiencies and fertilizer application cannot be addressed in detail within this research, the loss of nutrient-rich topsoil by soil erosion is provided here. It has been estimated that a large amount of nutrients are lost by soil erosion (Lal, 1988; Cogle et al., 2002). Therefore, an effort has been made to analyze the soil redistribution rates along the hillslopes. A quantitative method to measure the redistribution rates of soil erosion by a radionuclide tracer (^{137}Cs) and erosion modeling tools (WEPP model) to predict changes in soil loss due to land management practices are applied. The assessment of soil catenary development and topographically induced variability in soil properties served to confirm the catena concept and helped to differentiate the hillslope into several soil-landscape units, which are later considered by the erosion modeling approach.

Although it might not be possible to include all individual changes of soil characteristics along the slope in soil erosion modeling and even less in raster-based landscape modeling, it might be important to consider at least rough differences for five distinct slope units: summit, shoulder, backslope, footslope and toeslope/valley position.

3.5.3 Depth distribution of ^{137}Cs reference sample

The reference profile shows an exponential decline in ^{137}Cs concentration (Bq kg^{-1}) and ^{137}Cs areal activity (Bq m^{-2}) with increasing depth (Figure 3.8). The total concentration of ^{137}Cs is 10.1 ± 0.2 Bq kg^{-1} and total areal activity 729 ± 14.6 Bq m^{-2}. While most of the ^{137}Cs, approximately 75 %, is found in the upper 0-10 cm of the soil, it decreases to less than 7 % at a soil depth of 20-30 cm. The last depth increment (25-30 cm) holds only 3 % of the total measured concentration, but shows a high measuring error of 55 %. Compared with the low measurement errors in the upper soil (<6 % at 0-5 cm and <8 % at 5-10 cm), this high uncertainty value indicates the difficulty in detecting ^{137}Cs reliably below a depth of 25 cm. Deeper soil samples could not be taken because of an impermeable hard rock layer, which ensured, however, that no ^{137}Cs penetrated below a depth of 30 cm. Thus, all available ^{137}Cs was considered in this sampling.

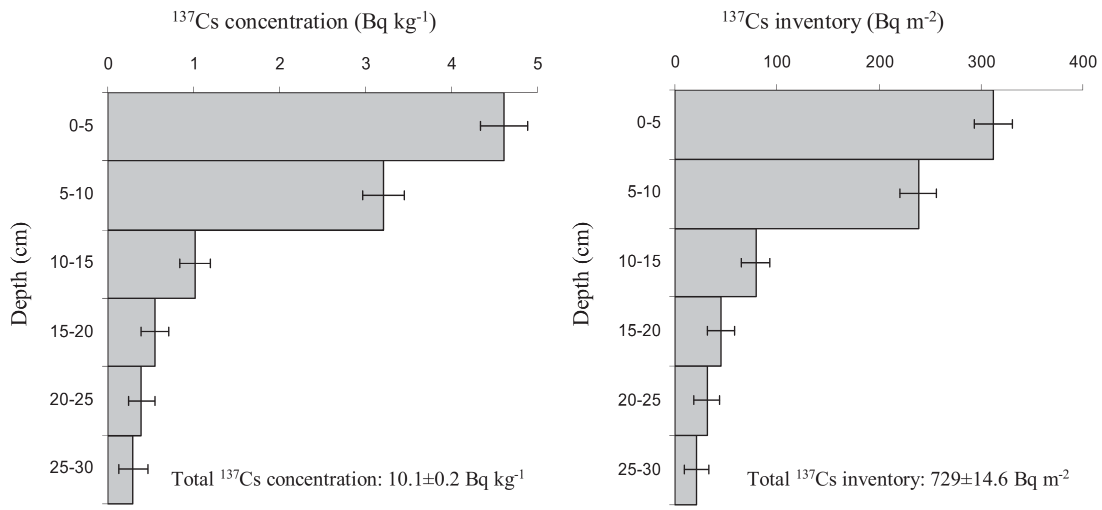


Figure 3.8 Depth profile of ^{137}Cs concentration and ^{137}Cs inventory at the reference site (Bontioli National Park, southwestern Burkina Faso)

The depth profile indicates a strong adsorption of ^{137}Cs in the upper soil horizons and only limited vertical migration, both characteristic for mineral soils, which have the capacity to adsorb and immobilize radionuclide fallout (He and Walling, 1996).

Furthermore, no downward displacement of the maximum ^{137}Cs peak is visible. Three assumptions can be made to explain the maximum peak of ^{137}Cs remaining in the upper 0-5 cm of the soil: Firstly, sampling increments of 5 cm thickness are not precise enough to reflect possible changes within a few centimeters such as a very low ^{137}Cs concentration in the upper 0-3 cm and the high ^{137}Cs concentration in the lower 3-5 cm. Secondly, the increase in biomass production and decomposition was very low, and the soil surface still reflects the topsoil layer of 40 years ago. Thirdly, the high activity of termites in this area has caused continuous soil redistribution in the upper soil by bioturbation processes, which has led to soil particle movement towards the surface. However, this ^{137}Cs depth distribution is confirmed by other studies in African environments, for example by Collins et al. (2001), who found a similar depth profile for undisturbed reference sites in southern Zambia with a maximum peak at 0-2 cm and 84 % of the total ^{137}Cs concentrated in the upper 10 cm of the soil.

Once a year, after the rainy season, small controlled bush fires are set in the protected area of Bontioli National Park. However, these small surface fires are considered to have only a minor impact on ^{137}Cs concentration in the soil. This is also confirmed by a study of Wallbrink et al. (2005), who assessed ^{137}Cs rates after severe bush fires and proved that ^{137}Cs losses were negligible, with a difference of 4 ± 1 % before and after the fire. However, as burning is also applied as a common practices on cultivated land after harvest, the reference inventory may still be used as representative, relative value.

The relatively low, but still significant ^{137}Cs inventory of 729 ± 14.6 Bq m^{-2} for Burkina Faso was compared with reference inventories from other sites in North, West and East Africa paying special attention to the geographical location, climate conditions and soil characteristics (Table 3.4). Although the geographical location and climate conditions are probably the most important factors influencing the distribution of ^{137}Cs worldwide, they alone might not suffice to explain the wide range of ^{137}Cs intensities in Africa. While the geographical location determines the distance to the source areas of nuclear-weapon testing, the climatic conditions and rainfall patterns control the intensities of radionuclide fallout. Global and local wind systems, climate regimes as well as variations in rainfall patterns will have a strong impact on ^{137}Cs fallout and might cause high local variations in ^{137}Cs rates.

Table 3.4 ^{137}Cs inventories for reference sites in North, West and East Africa

	Case study and authors	Geographical location	Average annual rainfall and climate conditions	Soil type and/ or parent material	^{137}Cs inventory for reference site (Bq m^{-2})
North Africa	Morocco Bouhlassa et al., 2000	Nakhla catchment, Rif region, northern Morocco, 35°28'N, 5°25'W, 160-1808 m asl.	660-800 mm, mediterranean sub-humid, two wet seasons; storms in summer	Lithosols, sandy-clay, clay-sand and clays	1671-3250
	Morocco Faleh et al., 2005	Abdelali catchment, Rif region, northern Morocco, 34°40'N, 3°52'W; 1160 m asl.	479 mm, semi-arid	Clay with gypsum, marls, limestones	370
West Africa	Niger Chappell et al., 1994	Fandou Béri, southwestern Niger, 13°32'N, 2°33'E, 240 m asl.	495 mm, semi-arid, two rainy seasons	Sandy loam, aeolian sands, sandy soils, low clay content	2517 ± 76
	Burkina Faso Schmengler (present study)	Bontioli National Park, southwestern Burkina Faso, 10°52'N, 3°04'W, 280 m asl.	955 mm, semi-arid, one rainy season	Ferric Lixisols on laterite plateau	729 ± 14
	Ghana Pennock, 2000	Kugri, northeastern Ghana, 10°45'N, 0°30'W 200 m asl.	1050 mm, semi-arid one rainy season	Alfisols, Alluvisols	925
East Africa (respective southern Africa)	Ethiopia Desta, 2005	Maidelle catchment, Tigray, northern Ethiopia, 13°11'N, 39°33'E, 2125 m asl.	500 mm, semi-arid one rainy season with minor rains in spring	Shale, limestone	1615
	Ethiopia Chekol, 2006	Rift valley, central Ethiopia, 8°60'N, 39°12'E, 1870 m asl.	900 mm, semi-arid one rainy season	Vertisol, mollic Andosol, silty clay loam	1253 ± 115
	Ethiopia, Argaw, 2005	Shomba catchment, southwestern Ethiopia, 7°23'N, 36°19'E, 1650 m asl.	1054-1820 mm, humid tropical one rainy season	Nitisols	2026 ± 176
	Uganda Rücker, 2005	a) Magada, Iganga District, southeastern Uganda, 0°32'N, 33°28'E, 1146 m asl.	1319 mm, humid tropical two rainy seasons	Plinthic Ferralsols, clay loam	439
		b) Kongta, Mount Elgon, eastern Uganda, 1°16'N, 34°45'E, 1819 m asl.	1259 mm humid tropical one rainy season	Mollic Andosols, clay to clay loam	392
	Zambia Collins et al., 2001	Kaleya catchment, southern Zambia, 16°11'S, 28°02'E, 1240-1400 m asl.	800-900 mm, moderate tropical, one rainy season	Ferruginous tropical laterites, limestones, calc-silicates	202.0 ± 22.8

However, ^{137}Cs concentrations diminish in general southwards. For example, ^{137}Cs inventories at 35°N in Morocco show values of 1671-3250 Bq m^{-2} (Bouhlassa, 2000), they decrease to 392-439 Bq m^{-2} around the equator in Uganda (Rücker, 2005) and reach further southwards at 16°S in Zambia only 202 Bq m^{-2} (Collins, 2001). A high variability of ^{137}Cs inventories was found in Ethiopia (Argaw, 2005; Chekol, 2006; Desta, 2005), which can be correlated neither with location nor with rainfall patterns. Site-specific characteristics such as soil properties, especially soil texture, might be the deciding factor for differences in ^{137}Cs.

Only a few studies are available to date focusing on ^{137}Cs reference values in West Africa and these mainly refer to ^{137}Cs redistribution rates in Niger (Chappell et al., 1994 and 1998; Chappell, 1999). The comparatively high ^{137}Cs values for southwestern Niger of 2517 ± 76 might be explained by the two rainy seasons there, which could have caused greater radionuclide fallout in the 1960s. Another explanation could be the high sand fraction of the soil. Sand-sized particles might adsorb high ^{137}Cs concentrations and - referring to the particle size experiments of He and Walling (1996)- ^{137}Cs concentrations of fine sediment eroded from a sandy soil might be much higher than those of fine sediments eroded from clay soils due to the higher radionuclide enrichment ratio of the sandy soil.

However, the ^{137}Cs study in northern Ghana by Pennock (2000) shows a reference value of 925 Bq m^{-2}, which corresponds well with the measured reference value for Burkina Faso in this study taking into account the geographical location at almost the same northern latitude and the slightly lower rainfall amount in Burkina Faso. Nevertheless, more ^{137}Cs studies are required to improve the very rough map of ^{137}Cs distribution in Africa and, additionally, more attention should be given to climate variations and soil characteristics, especially soil texture, which might have a determining influence on ^{137}Cs concentrations, especially when considering ^{137}Cs adsorption capacities and sediment enrichment ratios.

3.5.4 Soil redistribution patterns of ^{137}Cs

The local reference inventory was compared with ^{137}Cs samples collected from the two cultivated sites at Dano and Wahable in order to identify the spatial pattern of erosion and deposition zones at hillslope scale. Related to the reference inventory of

729 ± 14 Bq m^{-2} for a stable site, all ^{137}Cs values lower than the reference value indicate a loss of ^{137}Cs and hence net erosion, whereas all ^{137}Cs values higher than the reference value represent a gain and hence net deposition.

At Dano, the areal activity of ^{137}Cs ranges from 200 to 1500 Bq^{-2} and shows, in general, a continuous increase downslope (Figure 3.9). While the upper part of the hillslope experiences erosion – indicated especially by low ^{137}Cs inventories at shoulder position – the lower part of the hillslope receives deposition, indicated by high ^{137}Cs inventories at footslope and toeslope positions. The ^{137}Cs values at the middle part of the hillslope, mainly at the lower backslope position, are similar to those of the reference inventory. However, they do not represent stable, undisturbed conditions, but rather a balance between erosion processes at this point and sediment deposited from upslope areas. The hillslope does not show a distinct summit position, because the relief immediately drops in western direction after the elevation has reached its peak at the top of the hill. Besides the zonal distribution following contour lines, a few isolated ^{137}Cs peaks can be found at the shoulder and lower backslope position, where accumulation of ^{137}Cs sediments could have been caused either by small depressions in micro-relief or denser tree or bush vegetation.

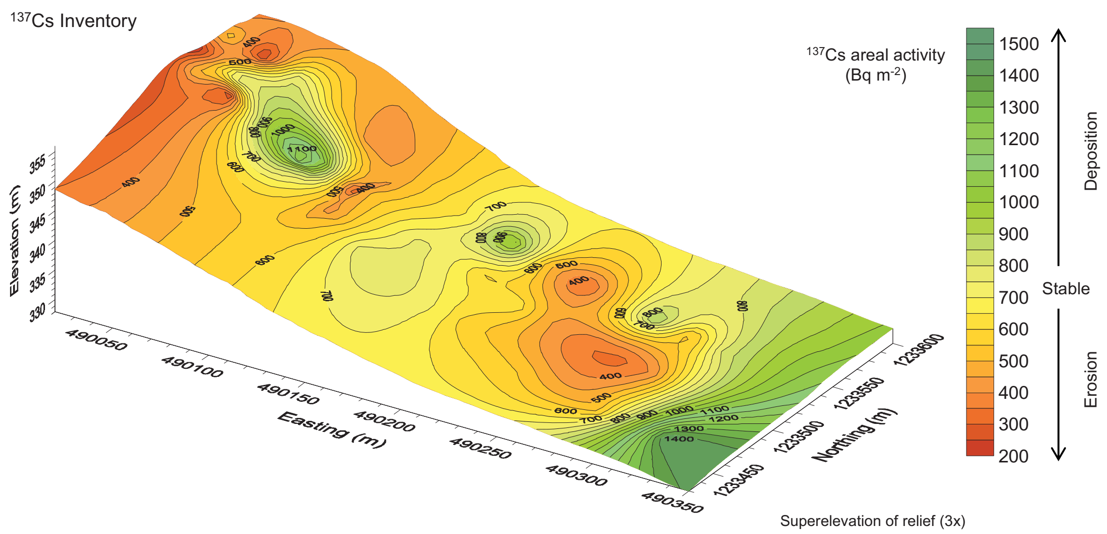


Figure 3.9 Spatial distribution of ^{137}Cs inventories at Dano hillslope, Burkina Faso

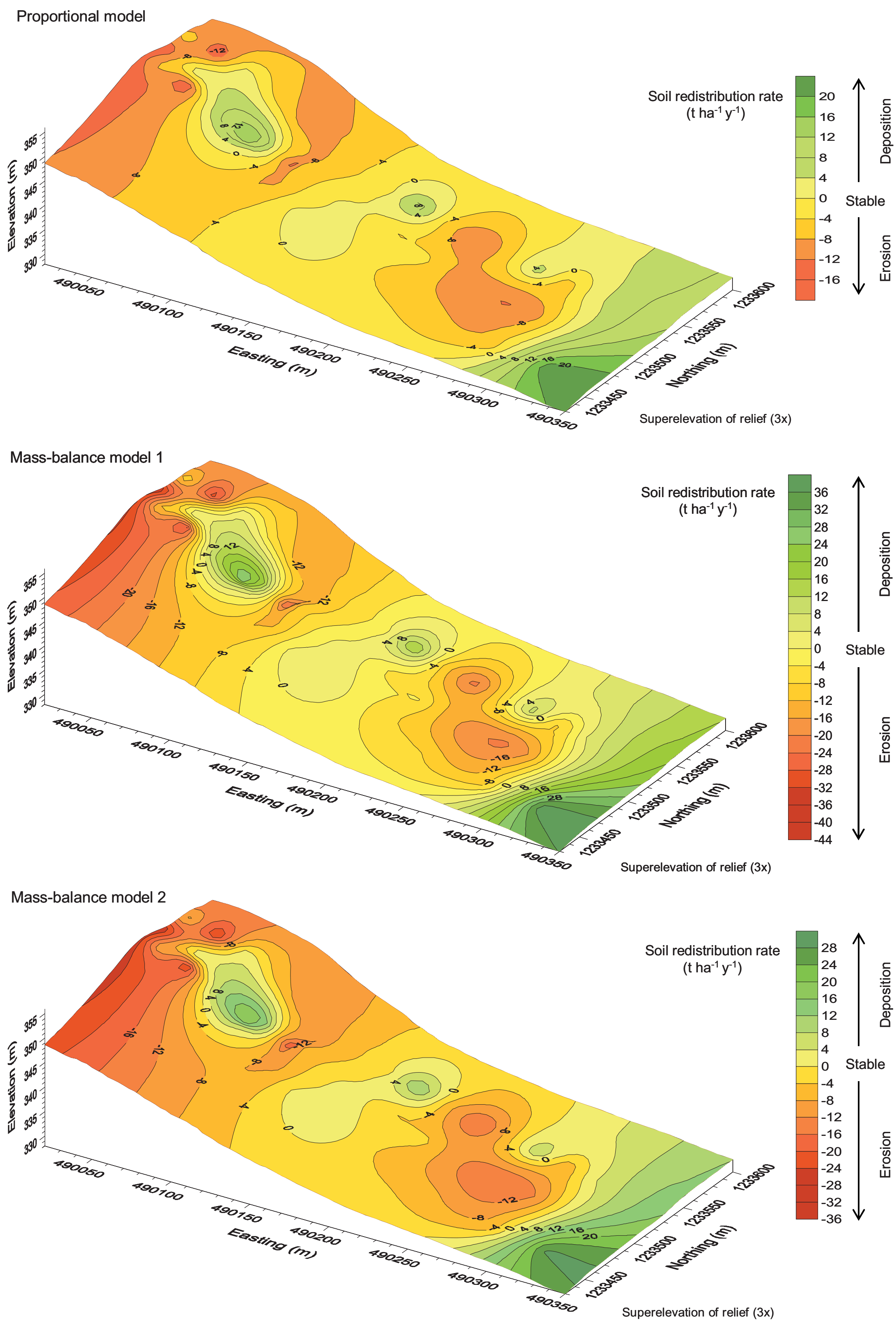


Figure 3.10 Spatial pattern of soil erosion and deposition at Dano Hillslope, Burkina Faso, based on different ^{137}Cs conversion models

In order to convert these relative ^{137}Cs inventories into quantitative soil erosion estimates, calibration procedures by conversion models are required. Therefore, the proportional model (PM) and two mass-balance models (MB-1 and MB-2) were used to calculate average annual soil erosion and deposition rates for the last 50 years (Figure 3.10). When comparing the pattern of soil redistribution for different conversion models, a remarkably high spatial agreement between erosion and deposition zones is observed, although the amplitude between maximum erosion and maximum deposition varies highly dependent on the conversion model used.

For example, the proportional model shows maximum redistribution rates of 18 t ha^{-1} yr^{-1} for erosion at shoulder position and 22 t ha^{-1} yr^{-1} for deposition at toeslope position, whereas the mass-balance models calculate for the same locations maximum erosion rates of 45 t ha^{-1} yr^{-1} (MB-1) and 37 t ha^{-1} yr^{-1} (MB-2), and maximum deposition rates of 38 t ha^{-1} yr^{-1} (MB-1) and 29 t ha^{-1} yr^{-1} (MB-2). Hence, the differentiation level increases when using mass-balance models, but the spatial distribution of erosion and deposition zones remains the same.

At Wahable, the areal activity of ^{137}Cs ranges from 100 to 2100 Bq m^{-2}, with lowest values at shoulder/backslope position indicating maximum erosion, and highest values at the footslope/toeslope position indicating maximum deposition (Figure 3.11).

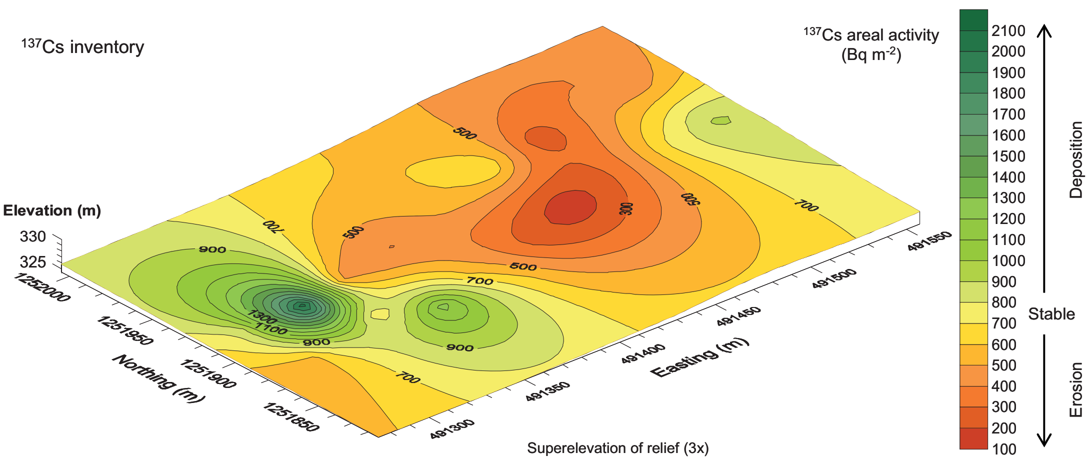


Figure 3.11 Spatial distribution of ^{137}Cs inventories at Wahable hillslope, Burkina Faso

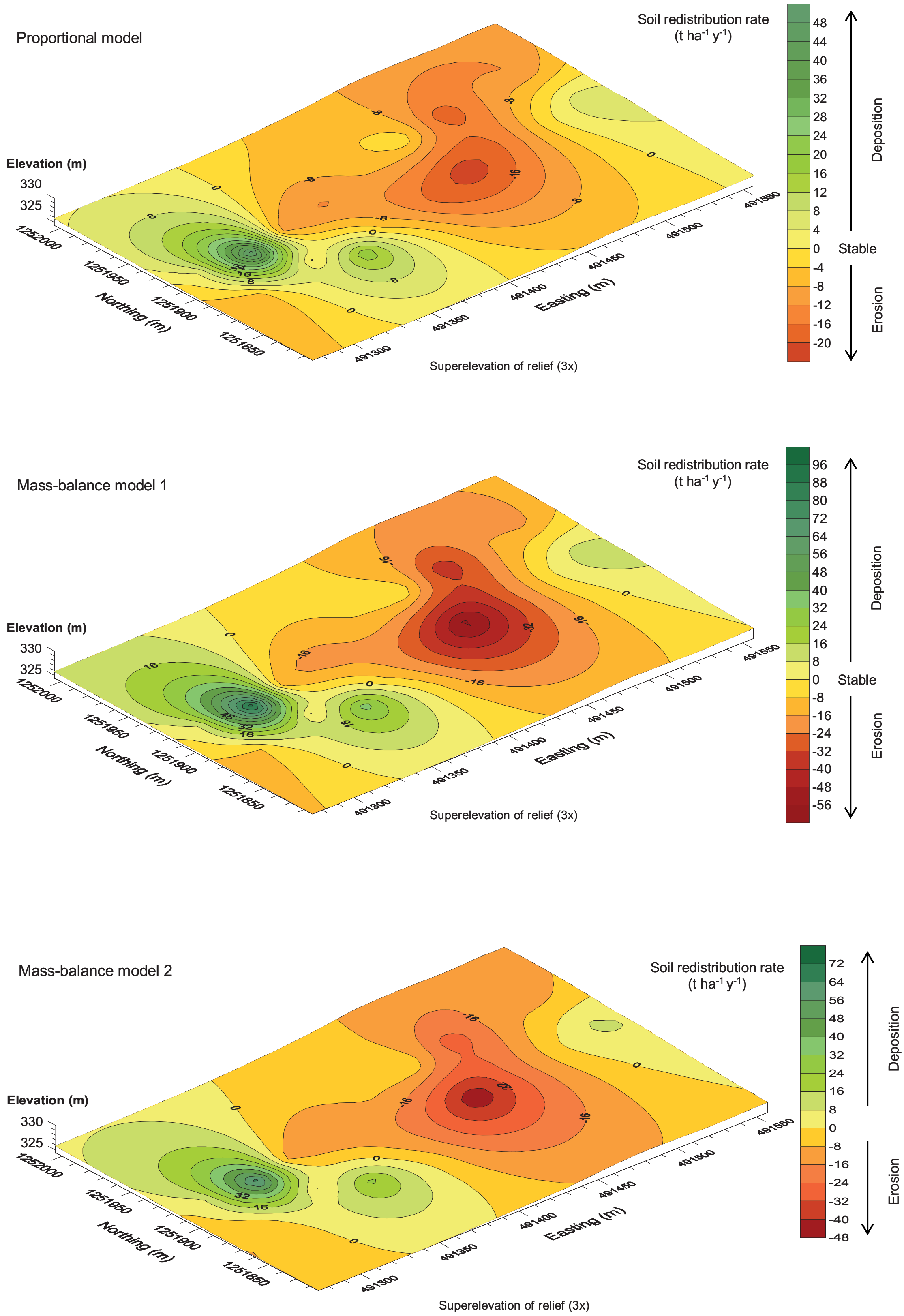


Figure 3.12 Spatial pattern of soil erosion and deposition at Wahable Hillslope, Burkina Faso, based on different ^{137}Cs conversion models

The almost stable ^{137}Cs values at the hilltop show that a true summit position has been reached. Additionally, almost stable values were measured at the lowest part of the hillslope below the maximum deposition zone, which might refer to a new summit position within the continuous landscape system. As the relief is less complex and the slope gradient very low (average slope gradients of 2.6°), the zonal pattern is pronounced with ^{137}Cs peaks falling within these zones.

Comparing the spatial pattern of soil erosion and deposition zones derived from conversion models, the high spatial correspondence in redistribution patterns is confirmed. Nevertheless, the differentiation level varies depending on the conversion model used (Figure 3.12). While the proportional model calculates maximum redistribution rates of 24 t ha^{-1} yr^{-1} for erosion at the shoulder position and 52 t ha^{-1} yr^{-1} for deposition at the footslope position, the mass-balance models show maximum erosion rates at these positions of 60 t ha^{-1} yr^{-1} (MB-1) and 48 t ha^{-1} yr^{-1} (MB-2), and maximum deposition rates of 96 t ha^{-1} yr^{-1} (MB-1) and 70 t ha^{-1} yr^{-1} (MB-2).

3.5.5 Comparison of soil erosion/deposition rates for two contrasting sites

Net erosion rates vary, depending on the conversion model used, between 2.4 and 3.3 t ha^{-1} yr^{-1} for the hillslope in Dano and between 2.7 and 5 t ha^{-1} yr^{-1} for the hillslope in Wahable (Table 3.5). The mass-balance model 1 produces the highest erosion at both sites. Average annual erosion rates calculated solely for the specific erosion areas are proportionally higher and range from 6.5 to 10.2 t ha^{-1} yr^{-1} at Dano, where erosion areas cover almost 70 % of the total area, and from 7.8 to 14.5 t ha^{-1} yr^{-1} in Wahable, where approximately 66 % of the total area is affected by erosion. Average annual deposition rates show a similar range when calculated for the specific depositional areas, but have lower gross deposition rates due to a smaller coverage of the total area. The sediment delivery ratio, a measure for the proportion of sediment eroded from the hillslope and transported to the valley, is approximately 50 % at both sites and varies only by about 3 % for different conversion models.

A comparison between both study sites shows that net erosion rates are very similar even though environmental conditions, in particular slope gradients, are very contrasting (see Table 3.2).

Table 3.5 Soil erosion/deposition rates derived from ^{137}Cs conversion models

Parameter	Proportional model		Mass-balance model 1		Mass-balance model 2	
	Dano	Wahable	Dano	Wahable	Dano	Wahable
Range of soil redistribution rate – max. erosion/deposition (t ha^{-1} yr^{-1})	-18.4 to 22.2	-22.9 to 51.8	-45.3 to 38.2	-59.9 to 96.1	-36.7 to 28.7	-47.5 to 70.1
Average erosion rate (t ha^{-1} yr^{-1}) (erosion areas)	-6.5	-7.8	-10.2	-14.5	-7.9	-10.9
Average deposition rate (t ha^{-1} yr^{-1}) (depositional areas)	7.0	7.2	11.4	13.2	8.7	10.2
Gross erosion rate (t ha^{-1} yr^{-1})	-4.5	-5.2	-6.9	-9.5	-5.4	-7.3
Gross deposition rate (t ha^{-1} yr^{-1})	2.1	2.5	3.6	4.5	2.8	3.4
Net erosion rate (t ha^{-1} yr^{-1})	-2.4	-2.7	-3.3	-5.0	-2.7	-3.8
Total soil loss (t yr^{-1})*	-25.5	-27.9	-38.9	-51.6	-30.3	-39.4
Total soil accumulation (t yr^{-1}) [(a)]	11.8	13.4	20.4	24.4	15.5	18.6
Sediment delivery ratio (%)	53.6	51.9	47.5	52.6	49.0	52.7

[(a)] *referring to a total area of 5.6 ha for Dano hillslope and 5.4 ha for Wahable hillslope*

Furthermore, the total amount of soil loss and soil accumulation is also within the same range at both sites and – unexpectedly - even higher on the flat hillslope of Wahable, especially when considering the slightly smaller surface area. Thus, topography could not be used as an explanation for the difference in the degree of erosion.

A reason for the apparent differences in soil erosion and deposition rates at the two sites may be related to the ^{137}Cs methods used. The clayey soils at Dano have a different effect on ^{137}Cs absorption capacity than the sandy soils in Wahable. In general, ^{137}Cs concentrations increase with an increase in the specific surface area and hence are generally much higher in the finer fraction (<0.063mm) than in the coarser fraction (Livens and Baxter, 1988; Walling and Woodward, 1992). However, some experiments have shown that also sand-sized particles may adsorb high ^{137}Cs concentrations, especially if the percentage of sand content in the samples is high, as it is the case for Wahable. Moreover, the radionuclide enrichment ratio, which is defined as “the ratio of the radionuclide concentration in the suspended sediment or eroded sediment to that in the original source material” (He and Walling, 1996, p. 127), can be much larger in sandy soils than in clayey soils. This would explain the high ^{137}Cs concentrations in deposited sediments from sandy soils and hence the high apparent deposition rates at Wahable.

One way to account for these differences in particle size is to adjust the particle size correction factor in the calculations. The particle size correction factor is one main input parameter required by all conversion models, and had to be set to 1.0 in all calculations due to limited data from field experiments. However, changing the particle size correction factor from 0.8 to 1.2, which represents a possible range, would lead to 10-20 % higher or lower erosion rates and thus would not completely explain the erosion rate differences in Dano and Wahable.

Another explanation for the relatively high values at Wahable, or rather the comparatively low erosion values at Dano, could be that the soil structure has a strong influence on soil erosion susceptibility. In general, soils with a high percentage of stones, like those in Dano, are very resistant to erosion, because a stone cover diminishes the detachment of soil particles - the so-called splash effect-, and prevents soils from drying out, reduces sediment supply, lowers sealing, and promotes biological activity. Thus, less soil material is transported downslope. Also the speed of runoff is slowed down by stones on the surface which cannot be transported due to their weight (Auerswald, 1998). At the same time, the high percentage of clay and humus in the soil increases the aggregate stability and reduces the detachment of particles. However, once detached, clay particles are also easily transported.

In contrast to the soil conditions at Dano, fine sand particles (0.063-0.1 mm) like those in Wahable are very susceptible to erosion. Their limited structural stability leads to a high degree of soil particle detachment. High rainfall intensities and a low vegetation cover at the beginning of the wet season will promote high runoff rates, which will be further aggravated by the process of sealing during detachment, and again enhance the transport of fine particles downslope.

Moreover, mechanical soil preparation techniques, such as ox-plough tillage at Wahable, may intensify soil erosion. The compression of the surface layer by the tillage implement will lead to a decrease in water infiltration capacity, an increase in surface sealing, and facilitated runoff generation and soil particle detachment (Poesen et al., 1986; Poesen and Lavee, 1994). Intensive tillage may, furthermore, reduce the aggregate stability and will have a de-stabilising effect due to the decomposition of organic matter (Auerswald, 1998). Therefore, one of the key parameters for erosion prevention will be the coverage of the soil surface by stones and/or vegetation, especially in periods of high rainfall intensities.

In this regard, soil conservation practices, such as stone lines along contours as found in Dano, may reduce erosion more effectively than contour ploughing by ox-plough at Wahable (see also section 3.5.8). Although the effect of stone lines, which have been piled up along contour lines since the mid 1980s (based on farmers knowledge), is not explicitly visible in the spatial pattern of ^{137}Cs distribution along the slope, it might have already had an overall positive impact on soil erosion rates.

Thus, a comparison between erosion rates at Dano and Wahable should be made with care, as the interpretation of ^{137}Cs results depends on many factors. These are on the one hand, various environmental characteristics, which might not be sufficiently incorporated in the ^{137}Cs approach, and on the other hand limitations and uncertainties due to ^{137}Cs measurements and conversion models. Also, more detailed field experiments, such as investigations on the ^{137}Cs absorption capacity of soil particles, are required to gain a better insight into ^{137}Cs redistribution processes.

One way to confirm some of the assumptions regarding tillage practices might be to use soil erosion models, such as the WEPP model, in order to simulate the influence of different land management options. Modelling the impact of ox-plough, handhoe or contour tillage would allow prediction of their impacts on soil loss rates at the two locations.

3.5.6 Correlation analysis of conversion models

A correlation analysis confirms the high correspondence among calculated results for each sampling point on the hillslope using the different ^{137}Cs conversion models. By comparing ^{137}Cs results of two models at a time, a positive correlation between all possible pairs of conversion models can be obtained (Figure 3.13). The highest correlation is found between the mass-balance model 1 and 2 (r^2=0.99), and the lowest between proportional model and mass-balance model 1 (r^2=0.89). Furthermore, the coefficient of correlation for each pair of conversion models is the same for both study sites. This might be a result of the linear internal equations of the conversion models and the consistency in input data for the individual site.

Variations in soil erosion estimates can also be explained by specific characteristics for each conversion model. The simplified proportional model, for example, has some obvious limitations: First of all, the model assumes a fixed time for ^{137}Cs enrichment of the soil and assumes that soil loss is directly proportional to the quantity of ^{137}Cs accumulation, although initial ^{137}Cs fallout started in the mid 1950s and continued over more than one decade. Secondly, the model does not consider that freshly accumulated ^{137}Cs might be removed by soil erosion before it could have been incorporated into the soil profile by tillage. Additionally, the model assumes that the ^{137}Cs amount is uniformly mixed within the plough layer (Walling and He, 2001). Thirdly, a fixed plough depth cannot account for surface lowering and continuous incorporation of subsoil into the plough layer, which would lead to an underestimation of soil loss. This was demonstrated by Walling et al. (2002), who showed that in some cases the proportional model underestimates erosion rates by more than 40 %. In our study, the proportional model also produced the lowest average soil redistribution rates for both sites, which were approximately 35-45 % lower than those of mass-balance model 1 and approximately 20-30 % lower than those of mass-balance model 2. Only the differences in net erosion rates for the hillslope at Dano were less, but were still 25 % for the mass-balance model 1 and 8 % for the mass-balance model 2.

These differences might be due to the more advanced structure of the simplified mass-balance model 1, which takes into account that the initial ^{137}Cs concentration in the upper soil layer is continuously reduced by ploughing processes in which soil from lower soil layers with negligible ^{137}Cs concentrations is mixed and incorporated into the original plough layer. However, mass-balance model 1 still suffers from the limitations mentioned above, such as a fixed time for ^{137}Cs fallout in the year 1963 and no possible removal of freshly accumulated ^{137}Cs fallout by heavy rainfalls before incorporation into the plough layer. Compared to the simplified mass-balance model 1, the improved mass-balance model 2 considers temporal variation of initial ^{137}Cs fallout such as initial infiltration depth of ^{137}Cs into the soil (relaxation depth) and its partial removal by heavy rainfalls before it was completely incorporated into the plough layer (proportional factor).

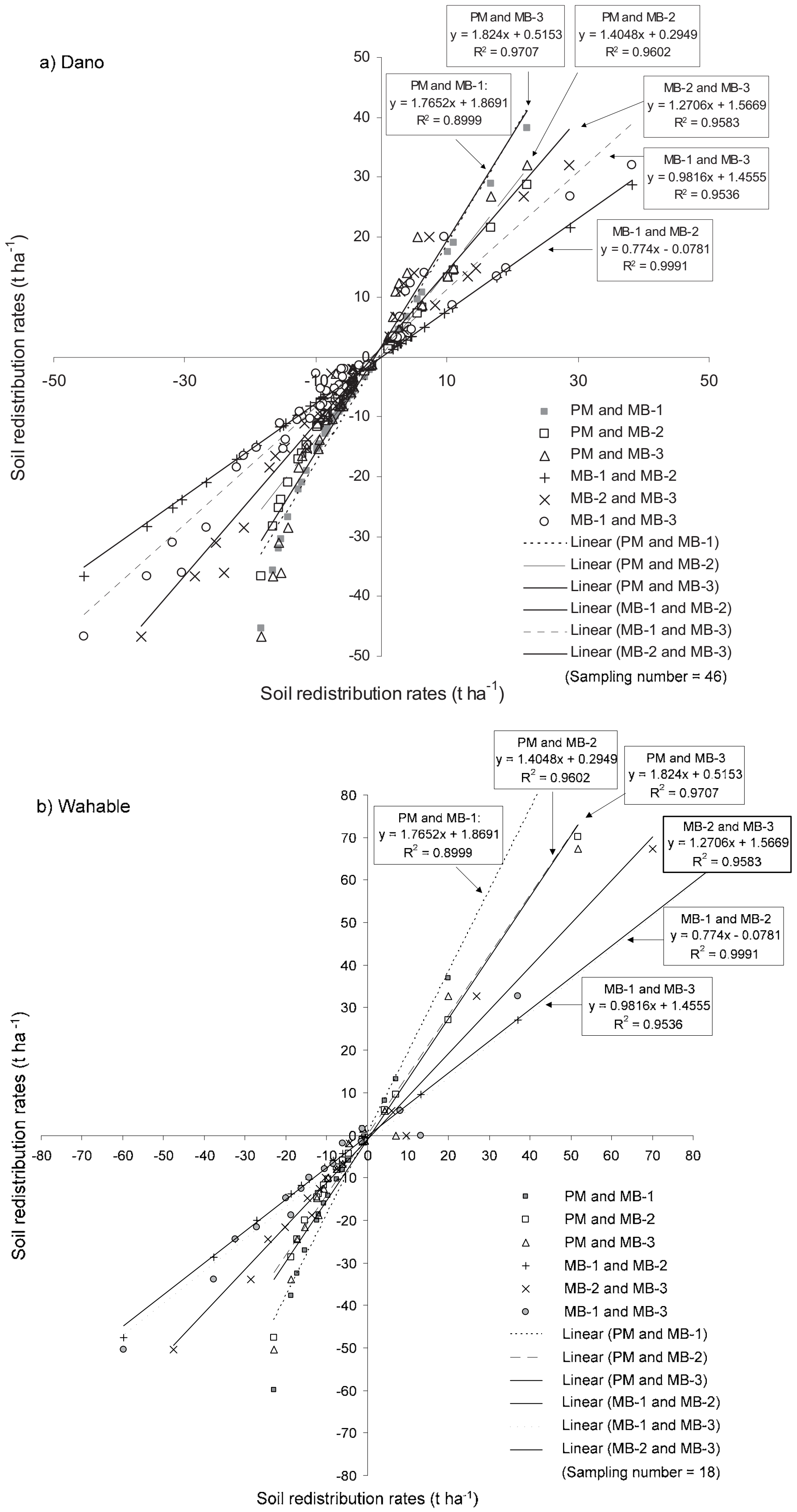


Figure 3.13 Correlation between pairs of conversion models in terms of soil redistribution rates for sampling points at two hillslope in Burkina Faso

Although the results might be more realistic than those of the simplified mass-balance model 1 (Walling and He, 2001), it is difficult to obtain the required input parameters, especially for regions like West Africa, where no experimental data about these values are available.

The mass-balance model 3 additionally takes the effects of tillage into account. In recent years, the effect of tillage on soil redistribution has been a main focus in many soil erosion studies (De Alba et al., 2004; Govers et al., 1999; Lindstrom et al., 1992; van Oost et al., 2000), and its importance has been also emphasized for the ^{137}Cs approach (Li et al., 2007; Lobb et al., 1995; Quine et al., 1999; Schumacher et al., 2005; van Oost et al., 2003; Walling and Quine, 1993). As topography is one of the most important factors controlling soil redistribution by tillage, slope gradients along a catena serve as input parameters to calculate simultaneously redistribution rates by tillage as well as redistribution rates by water erosion. However, soil redistribution by tillage plays a minor role in Burkina Faso - in comparison to more mechanized agriculture in developed countries - as hand-hoe or ox-plough tillage will cause only slight soil disturbance.

In summary, the results of the correlation analysis show that – in this case - differences in ^{137}Cs results due to the choice of conversion models are very low and cannot explain differences in soil erosion rates at the study sites in Burkina Faso. In other areas, however, they might have a stronger influence on the range of soil redistribution rates, especially when soil redistribution by tillage plays a more dominant role.

3.5.7 WEPP-simulated soil loss and soil deposition

Simulated annual soil loss as calculated by the WEPP model shows both high erosion rates along transect lines and high differences in soil loss rates for individual hillslope positions at the hillslope in Dano (Figure 3.14). Whereas average annual soil loss rates for the entire hillslope are 35.2 t ha^{-1} y^{-1} for transect 1 and 66 t ha^{-1} y^{-1} for transect 2, variations in soil loss range from 4.7 t ha^{-1} y^{-1} at summit positions to a maximum of 211.6 t ha^{-1} y^{-1} at the steeper backslope positions. Additionally, soil deposition reaches a peak of 29.2 t ha^{-1} y^{-1} at the footslope and 12.9 t ha^{-1} y^{-1} at the toeslope/valley positions.

For both transect lines, the pattern of soil loss increases with the highest soil loss at the lower backslope position and the highest deposition rates at the footslope position. Absolute soil loss values vary significantly, especially at the lower backslope position where a difference in soil loss of more than 100 t ha^{-1} y^{-1} is observed. This significant difference can be explained by the high sensitivity of the WEPP model to changes in soil properties, especially soil texture, which has a strong influence on the model's internally calculated values for soil bulk density, porosity, water retention and hydraulic conductivity (Alberts et al., 1995; Flanagan and Livingstone, 1995). Input parameters for soil and slope properties differ between transects due to individual field measurements whereas climate conditions, land management and tillage operations stay the same (Table 3.6). Slope gradients, which are known to have a decisive impact on soil erosion simulations, are slightly steeper for the second transect, but do not account for the large differences in soil loss. For example, slope gradients of the second transect are steeper at the shoulder position (OFE 2 and OFE 3) but simulated soil loss is lower, and slope gradients are slightly lower at the backslope position but simulated soil loss is higher.

Therefore, another explanation is needed, and hence the model's sensitiveness to soil erodibility, critical shear stress and saturated hydraulic conductivity is called for. These factors are known as the dominant parameters influencing the susceptibility or resistance to erosion of the soil (Elliot et al., 1990; Kuhn, 2006; Nearing et al., 1990). A sensitivity analysis has shown that the model is highly responsive to changes in soil characteristics (Brunner et al., 2004; Schröder, 2000), and that especially erodibility parameters have a major impact on simulated soil loss (Nearing et al., 1990).

The WEPP model calculates erodibility parameters separately for rill areas, which are characterized by concentrated flow paths along the slope, and interrill areas, which are characterized by overland flow between these rills. Whereas rill erodibility (Kr) describes the detachment and transport of suspended sediment by concentrated flow as a function of the soil's susceptibility, critical shear stress and sediment transport capacity, interrill erodibility (Ki) represents the detachment of soil particles by the impact of raindrops (splash effect) and thus expresses the sediment delivery ratio into rills as a function of rainfall intensity and runoff rate (Alberts et al., 1995).

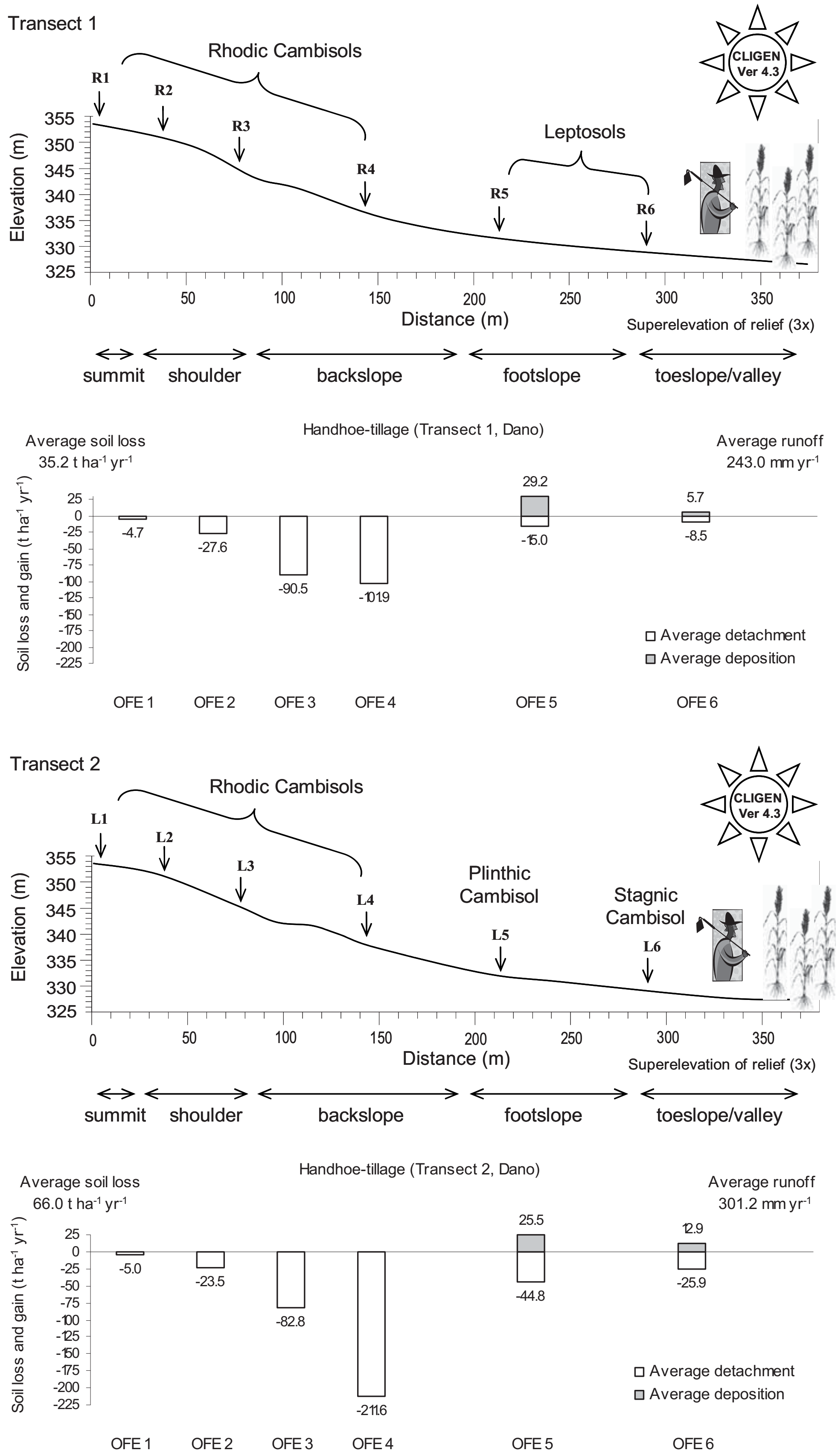


Figure 3.14 Simulated soil loss along transects at Dano hillslope, Burkina Faso

Table 3.6 Input parameters for simulation by WEPP model for Dano, Burkina Faso

WEPP model Input parameter (Hillslope of Dano)	**Hillslope section = Overland Flow Element (OFE)**											
	OFE 1		OFE 2		OFE 3		OFE 4		OFE 5		OFE 6	
	T1	T2	T1	T2	T1	T2	T1	T2	T1	T2	T1	T2
Slope gradient (%)	5.6	9.4	11.2	16.4	13.4	14.2	11.6	9.8	4.0	4.4	3.7	4.3
OFE length (m) Accumulated length	30 30	30 30	30 60	30 60	60 120	60 120	60 180	60 180	90 270	80 260	90 360	80 340
Soil texture [(a)] (clay content %; silt content %; sand content %)	C (42 40 18)	C (59 38 3)	C (49 36 15)	C (59 34 7)	SC (44 42 14)	C (60 35 5)	CL (31 47 22)	SC (42 45 13)	C (48 37 15)	C (51 34 15)	CL (30 39 31)	CL (33 39 28)
Interrill erodibility [(b)] (Ki) (10^6 kg s m^{-4})	0.1– 0.2	0.1- 0.2	0.1- 0.2	0.1- 0.2	0.1- 0.2	0.1- 0.2	0.1- 0.2	0.1- 0.2	0.1	0.1	0.1	0.1
Rill erodibility [(b)] (Kr) (10^3 s m^{-1})	0.3- 1.5	0.2- 0.3	0.2- 1.0	0.2- 0.3	0.2- 1.1	0.2- 0.7	0.3- 1.7	0.5- 1.9	0.2- 0.9	0.5- 1.9	0.2- 0.6	0.4- 1.8
Effective saturated hydraulic conductivity (Ke) [(b)] (mm h^{-1})	0.8- 4.0	0.2- 3.0	0.2- 3.0	0.2- 3.0	0.6- 3.0	0.2- 3.0	1.0- 3.5	0.4- 3.0	0.5- 3.0	0.4- 3.0	1.0- 5.0	0.5- 5.0
Climate	Climate Generator (CLIGEN, Ver. 4.3); >50y simulations (1957-2006)											
Land management	Plant: *Sorghum Bicolor*											
Tillage operations	Ploughing by handhoe											

[(a)] *Soil texture of A-horizon*
[(b)] *Internally calculated and adjusted by WEPP model within continuous simulation*

Interrill erodibility ranges at both transects between 0.1 and $0.2*10^6$ kg s m^{-4} and decreases slightly at footslope/valley positions (OFE 5 and OFE 6) as a result of lower slope gradients and hence reduced runoff rates. Rill erodibility varies between 0.2 and $1.9*10^3$ s m^{-1} and shows an increase at the lower backslope position (OFE 4), which can be related to the higher silt content, and might thus explain the higher soil loss rates. In general, soils with a higher silt fraction are more susceptible to detachment and transport and hence show higher soil erodibility (Wischmeier and Mannering, 1969; Elliot, 1990). At both transects, rill erosion is the dominant source for eroded sediment and decisive for the high soil loss rates. Soil detachment by interrill erosion contributed only 5-10 % of the total soil loss. The calculated erodibility values can be interpreted as very low when compared with adjusted interrill erodibility values of 0.7 to $4.1*10^6$ kg s m^{-4} and adjusted rill erodibility values of 0.6 to $45.3*10^3$ s m^{-1} for cropland soils in the USA (Elliot et al., 1990).

Effective saturated hydraulic conductivity (Ke), which is a key parameter controlling infiltration rates and runoff during rainstorms based on the Green and Ampt equation (Green and Ampt, 1911; Alberts, 1995), was also internally calculated and adjusted by the WEPP model, and shows values between 0.2 and 5 mm h^{-1}. These values correspond with recommended hydraulic conductivity values of 0.6 mm h^{-1} by Rawls et al. (1982) and 2.0 mm h^{-1} by Clapp and Hornberger (1978) for clayey soils. The very low hydraulic conductivity is mainly caused by soil's high bulk density and high clay content which restricts infiltration. Runoff occurs when the precipitation amount exceeds the infiltration capacity of the soil and its surface storage is filled. This condition is usually reached more quickly in soils with a high percentage of clay and a low percentage of sand, because finer clay particles impede raindrop infiltration and tend to compact and consolidate faster than sandy or loamy soil particles. Therefore, clayey soils generate a higher amount of runoff than sandy soils. The high rainfall intensity in tropical environments may accelerate this effect.

In terms of soil erodibility, most erosion studies refer to the erodibility factor (K) of the USLE, which expresses erodibility as related to soil texture, organic matter content, soil structure and permeability (Wischmeier and Schmith, 1978). Therefore, soil erodibility was additionally determined by the soil-erodibility nomograph to compare model-calculated erodibility values (Kr) with standard USLE estimates (K) and measured data from field experiments. The estimated erodibility factor K for the study site ranged from 0.2 to 0.3, with highest values at the backslope position (OFE 4), which is in agreement with calculated rill erodibility values (Kr). Compared with field measurements in West Africa, these values indicate a moderate soil erodibility, although in a more global context they can be interpreted as relatively low. A wide range of field studies confirms that tropical soils in West Africa are generally less erodible (K between 0.001 to 0.4) than soils in temperate zones (K between 0.03 to 0.7) (El Swaify et al., 1982; Roose and De Noni, 2004; Roose and Sarrailh, 1990). In Burkina Faso, estimated K values vary between 0.01 and 0.28 for cambisols in the south-Sudanien zone (Roose and Piot, 1984), but can reach values of up to 0.45-0.59 on cambisols in northern Burkina Faso (Mietton, 1988) where infiltration is restricted by low soil depth and a subsoil laterite layer (Roose and Sarrailh, 1990).

Considering adjusted erodibility parameters, the WEPP model seems to underestimate interrill and rill erodibility. This might be due to the model's limitation in reflecting the effect of mineralogy and aggregate stability by using a simple integration of texture classes in percentage. Model sensitivity in terms of clay content might be the dominant factor leading to low rill erodibility rates. It accounts for the cohesion characteristics of clay particles that represent the main force in resisting soil detachment. Additionally, the model responds to the high organic matter content (2-4 %) and the high bulk density values (1.1-1.9 g cm^{-3}) of the surface layer, which also have an important effect on soil erodibility. However, as the susceptibility of a soil to erosion is spatially and temporally highly variable and depends on the interaction of multiple factors, a clear relationship between soil properties and soil erodibility cannot be derived without further intensive field studies, to allow adequate parameter adjustment and model calibration to tropical conditions.

At the hillslope of Wahable, which is characterized by lower slope gradients and sandy loam Lixisols (Table 3.7), simulated average soil loss is comparatively lower and amounts to 5 t ha^{-1} y^{-1} for the entire transect 1 and 1.5 t ha^{-1} y^{-1} for transect 2 (Figure 3.15). Soil loss rates show a very similar pattern at individual hillslope positions and differ only in simulated soil deposition rates at the toeslope/valley position, where a four times higher amount of deposition is reached at transect 2. This difference is mainly caused by the slightly inclined hillslope towards transect 2 leading to an increased soil accumulation from upslope sediment at this lower hillslope area. Highest soil loss rates of 8.7 to 10.6 t ha^{-1} y^{-1} are reached at the backslope positions, which corresponds well with comparatively steepest slope gradients of 4.4 to 4.8°%.

Whereas interrill erodibility is approximately $0.2*10^6$ kg s m^{-4} for both transects and shows no significant variation along the hillslope, rill erodibility ranges from 0.3 to $1.8*10^3$ kg s m^{-1} and increases slightly but not significantly at the backslope position. Rill erosion represents also here the dominant source for eroded sediment. Sediment contribution from interrill erosion is negligibly low. Both erodibility values are within the same range of those of the hillslope in Dano, although the sandy loamy soil texture at Wahable would usually imply a higher susceptibility to erosion and hence a higher soil erodibility.

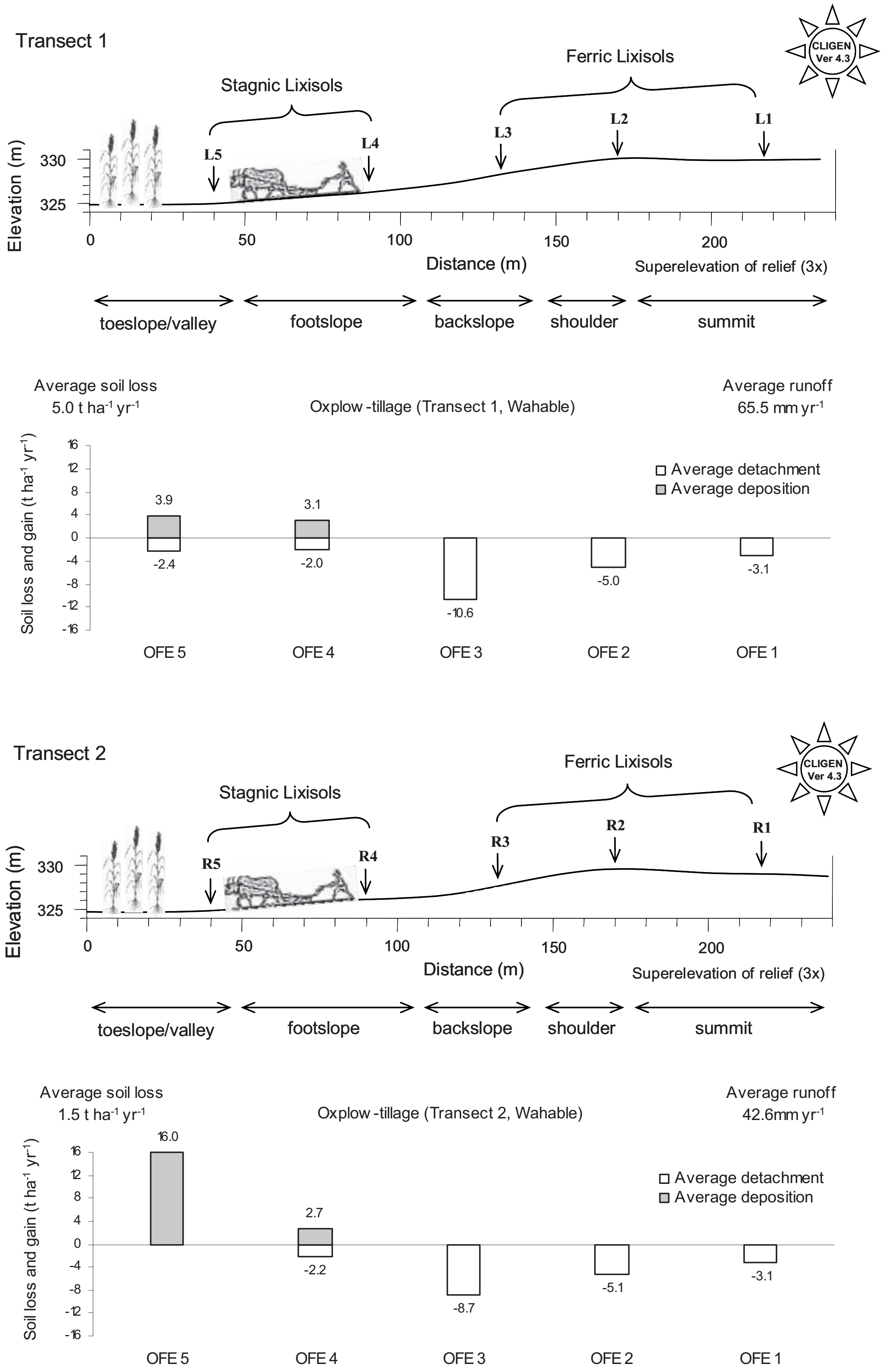


Figure 3.15 Simulated soil loss along transect at Wahable hillslope, Burkina Faso

Table 3.7 Input parameters for simulation by WEPP model, Wahable, Burkina Faso

WEPP model Input parameter (Hillslope of Wahable)	**Hillslope section = Overland Flow Element (OFE)**									
	OFE 1		OFE 2		OFE 3		OFE 4		OFE 5	
	T1	T2	T1	T2	T1	T2	T1	T2	T1	T2
Slope gradient (%)	1.7	2.9	2.6	3.3	4.8	4.4	2.4	2.4	1.9	1.2
OFE length (m) Accumulated length	50 50	50 50	50 100	50 100	50 150	50 150	50 200	50 200	35 235	35 235
Soil texture [a] (clay content %; silt content %; sand content %)	SL (11 27 62)	SL (89 30 61)	SL (12 22 66)	SL (16 28 56)	SL (11 26 63)	SL (17 26 57)	SL (8 25 67)	SL (8 29 63)	L (9 40 51)	SL (9 35 66)
Interrill erodibility [b] (Ki) (10^6 kg s m^{-4})	0.2	0.2	0.2	0.2	0.2	0.2	0.1- 0.2	0.2	0.2	0.2
Rill erodibility [b] (Kr) (10^3 s m^{-1})	0.8- 1.8	0.4- 0.6	0.5- 1.2	0.4- 0.9	0.5- 1.4	0.4- 1.0	0.5- 1.2	0.5- 1.1	0.3- 0.6	0.4- 0.7
Effective saturated hydraulic conductivity (Ke) [b] (mm h^{-1})	9.0- 22.0	9.0- 21.0	8.0- 22.0	5.0- 15.0	7.0- 19.0	5.0- 15.0	10.0- 23.0	9.0- 21.0	7.0- 14.0	9.0- 21.0
Climate	Climate Generator (CLIGEN, Ver. 4.3); >50y simulations (1957-2006)									
Land management	Plant: *Sorghum Bicolor*									
Tillage operations	Ploughing by handhoe									

[a] *Soil texture of A-horizon*
[b] *Internally calculated and adjusted by WEPP model within continuous simulation*

The relatively low values might be explained by the model's response to slope gradients (1.2-4.8 %) and hence low critical shear stress as well as by the reduced organic matter content (0.4-1.5 %) and comparatively low bulk density values (1.4-1.6 g cm^{-3}). These are all known to be negatively related to soil particle detachment (Alberts et al., 1995; Romero et al., 2007). Additionally, infiltration rates on sandy loamy soils are comparatively higher than on clayey soils, thus leading to reduced soil loss rates. Effective saturated hydraulic conductivity varies between 5 and 23 mm h^{-1} and is slightly below the recommended value of 26 mm h^{-1} (Rawls et al., 1982) and 44 mm h^{-1} (Clapp and Hornberger, 1978) for sandy loamy soil, but still within the range of measured values for the same soil texture characteristics in the savannah zone of Northern Ghana, which vary between 5 and 37 mm h^{-1} (Kpongor, 2007; Agyare, 2004).

When estimating soil erodibility with the USLE nomograph, the lixisols of Wahable can be classified as moderately to highly erodible with factor values between 0.25 and 0.4. This factor range corresponds with general estimations of 0.2 to 0.3 (Roose and De Nooni, 2004) for ferric lixisols in West Africa, which are known to be relatively fragile and might become even more susceptible to erosion after a long time of cultivation due to a decrease in organic matter content and decreased aggregate stability (Roose and Sarrailh, 1990).

A comparison between both study sites shows that annual net erosion rates at Dano are more than 10-fold those at Wahable (Table 3.8). Although average and gross erosion rates show the same amplitude of difference between both sites, average and gross deposition rates are almost within the same range of magnitude and only slightly higher at Dano when calculated solely for the depositional areas. Maximum erosion and deposition rates are strongly related to slope gradients along the transect lines.

Table 3.8 Soil erosion/deposition rates simulated by WEPP model

WEPP –simulation	Dano		Wahable	
	Transect 1	Transect 2	Transect 1	Transect 2
Average annual runoff (mm)	243.0	301.2	61.9	42.6
Average erosion rate (t ha^{-1} yr^{-1}) (erosion length in % of total transect)	-44.6 (87 %)	-74.1 (92 %)	-5.1 (90 %)	-4.9 (82 %)
Average deposition rate (t ha^{-1} yr^{-1}) (deposit. length in % of total transect)	26.9 (13 %)	21.3 (8 %)	3.2 (10 %)	13.6 (18 %)
Gross erosion rate (t ha^{-1} yr^{-1}) [a)]	-38.7	-67.8	-4.6	-4.0
Gross deposition rate (t ha^{-1} yr^{-1}) [a)]	3.5	1.8	0.4	2.5
Net erosion rate (t ha^{-1} yr^{-1})	-35.2	-66.0	-4.2	-1.5
Max. erosion (t ha^{-1} yr^{-1}) (point along transect, m)	-149.1 (at 132.0)	-320.9 (at 121.0)	-12.3 (at 100.0)	-14.8 (at 117.5)
Max. deposition (t ha^{-1} yr^{-1}) (point along transect, m)	61.6 (at 229.0)	61.0 (at 222.0)	9.7 (at 200.0)	19.2 (222.0)
Sediment delivery ratio (%)	90.8	97.4	92.8	38.6

[(a)] *referring to a total length of 360 m for transect 1 and 340 m for transect 2 at Dano hillslope and a total length of 235 m for transect 1 and 2 at Wahable hillslope; positive values indicate soil gain, negative values soil loss.*

At both study sites, approximately 80-90 % of the area is affected by erosion and proportionally little by deposition, even though simulated deposition rates might reach up to 62 t ha y^{-1} at Dano and almost 20 t ha y^{-1} at Wahable. The sediment delivery ratio is above 90 %, which indicates that most of the sediment is transported further downslope without being accumulated or intermediately stored along the hillslope. An exception is the second transect of Wahable, where a pronounced valley position functions as a depositional storage for approximately 60 % of the upslope sediment.

The great differences in simulated soil loss between both sites might be explained firstly by topographical conditions, such as higher slope gradients at Dano, and secondly by the highly distinct soil properties, such as soil texture, organic matter content and hydraulic conductivity, which have a significant effect on simulation results. For example, by assuming sandy loamy lixisols at the hillslope of Dano leaving all other parameters unchanged, net erosion rates would decrease to 3.7 t ha y^{-1} (which is approximately 10 % of the originally simulated net erosion rate) for the first transect and to 3.9 t ha y^{-1} (6 %) for the second. In contrast, assuming rhodic cambisols at Wahable, soil loss rates would increase to 16.7 t ha y^{-1} (394 %) for the first transect and to 6.1 t ha y^{-1} (397 %) for the second. On the other hand, by exchanging slope parameters and assuming lower slope gradients at Dano, net erosion rates would decrease to 6.3 t ha y^{-1} (18 %) and 8.3 t ha y^{-1} (12.6 %) at Dano, whereas higher slope gradients at Wahable would lead to increased net erosion rates of 8.2 t ha y^{-1} (195 %) and 8.9 t ha y^{-1} (576 %), respectively. This indicates that changes in soil properties have an impact on simulated soil erosion rates that is similarly high to that of changes in slope characteristics.

Erosion rates between 10 and 200 t ha^{-1} y^{-1} are reported as typical for the savannah ecosystems (Mati and Veihe, 2001). Measured data from experimental stations in West Africa (Table 2.1, Chapter 2) show that soil erosion rates under similar climate conditions (500-1300 mm rainfall) usually range from 0.1 to 26 t ha^{-1} y^{-1} on cultivated soils with slope gradients between 0.5 and 4 %, but might reach up to 85 t ha^{-1} y^{-1} on leached, sandy clay soils with slope gradients of 4 % (Roose, 1976 and 1994).

However, although soil erosion rates simulated by the WEPP model are within the range of measured data, absolute soil loss data should be considered with care. As model predictions have shown, simulated soil loss varies significantly not only between different landscape units, here presented in form of two pedo-geomorphically distinct hillslopes, but also between parallel soil transects located at the same hillslope only a few ten meters apart from each other. Nevertheless, the pattern of simulated soil loss at all sites is similar, confirming the reciprocal relationship between catenary soil development and erosion processes. This suggests that each landscape position responds differently to erosion processes along the hillslope. Simulated soil loss is most severe at shoulder and backslope position, which corresponds with pedo-geomorphological findings (e.g., higher slope gradients, slope convexity, thinner A-horizons) indicating that these landscape positions are related to lateral soil water movement and intensive sediment transport downslope. Soil accumulation, in return, predominates at footslope and toeslope/valley positions, which agrees with slope geometry and diagnostic soil profile characteristics at these positions (e.g., low slope gradients, straight to slightly concave slope geometry, thicker A-horizons). This is where colluvial redepositions of sediment from uplslope are found. Therefore, simulations considering the catenary soil sequence in their modeling approaches, e.g., by dividing the hillslope into distinct soil-landscape units, can better account for the variability of soil and hence show a clear spatial demarcation between erosion and sedimentation zones. Furthermore, the integration of the catenary concept provides a promising framework to adjust land management methods to the specific requirements of individual hillslope sections.

3.5.8 Soil erosion scenarios under different land management options

The effect of soil and water conservation techniques (e.g., stone lines, contour farming) as well as actual and potential land management options (e.g., handhoe, ox-plough and minimum tillage, residue addition, burning and tillage by chisel plow) is assessed by comparing simulated soil loss for each individual hillslope unit (OFE) along a representative transect at each study site (Figure 3.16 and 3.17).

The WEPP model simulations indicate first of all, that the magnitude of soil loss and soil accumulation varies significantly along the hillslope depending on the land management practice, and secondly, that the effect of each practice is different at individual OFEs, although a consistent pattern due to hillslope positions can be observed. Thus, some methods might be more appropriate for reducing soil loss at specific erosion-prone hillslope areas than others.

At the hillslope in Dano, commonly used conservative management practices, such as tillage by handhoe, minimum tillage and residue management, generate soil loss rates of 4.1 t ha^{-1} y^{-1} at summit position (OFE 1) and 101.9 t ha^{-1} y^{-1} at backslope position (OFE 4).

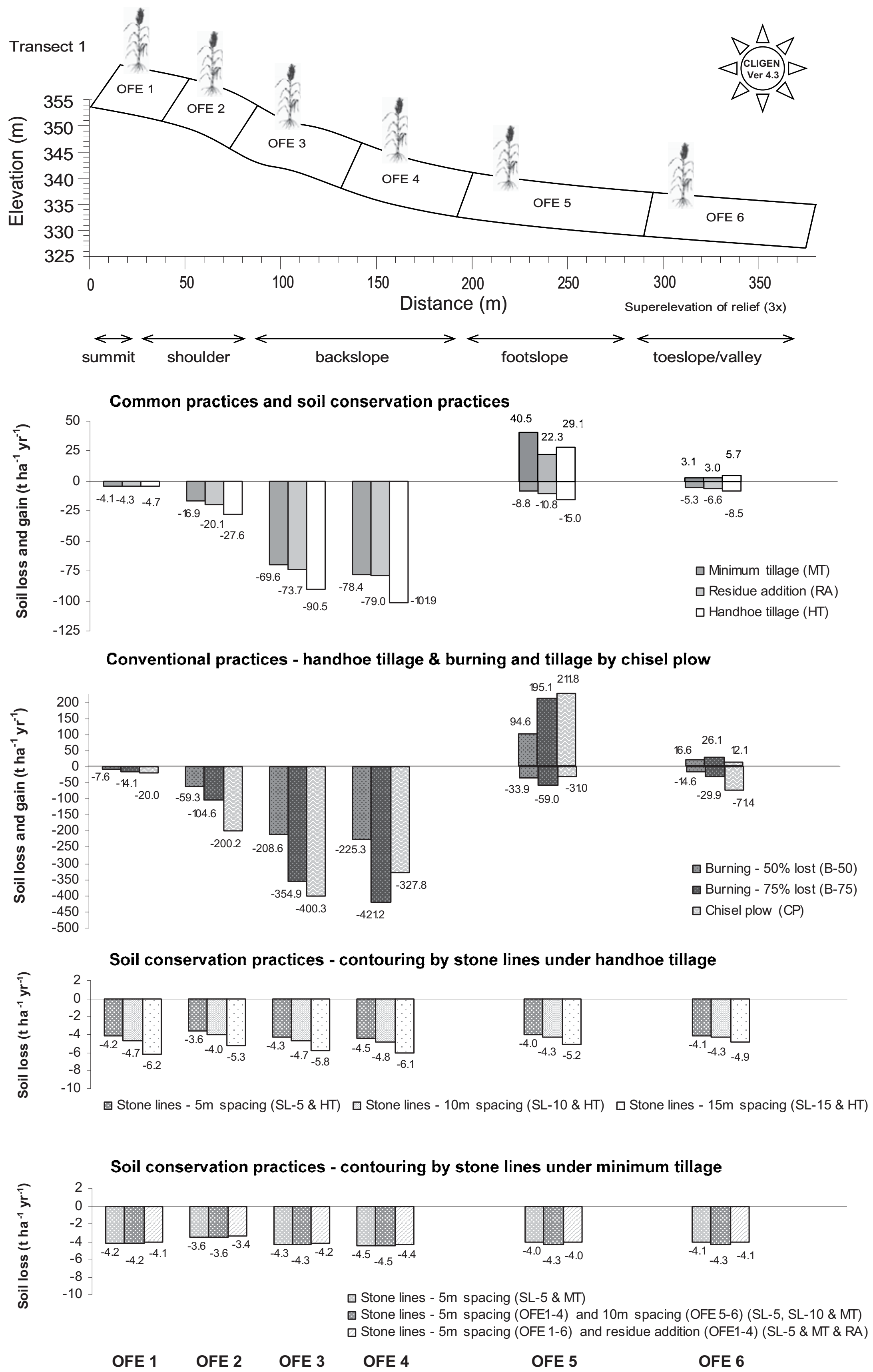


Figure 3.16 Effect of land management options on simulated soil loss at Dano

This is three to four times lower than the losses with conventional burning practices or tillage by chisel plough, which can vary between 7.6-20.0 t ha^{-1} y^{-1} (OFE 1) and 225.3-421.2 t ha^{-1} y^{-1} (OFE 4), respectively (Figure 3.16). The increase in soil loss under high-tech tillage is assumed to be a result of high soil surface disturbance and soil compaction due to the mechanical tillage implement. In contrast, minimum tillage causes a low surface disturbance and is the most effective option to reduce soil loss.

Furthermore, the assumption that soil conservation practices, such as stone lines along contours, have a significant impact in preventing soil loss is confirmed by simulations. Soil erosion rates range from 3.4 t ha^{-1} y^{-1} to 6.2 t ha^{-1} y^{-1} for stone line applications and show no significant variations between individual slope positions. Stone lines represent relatively solid runoff and soil loss barriers and divide the hillslope into several smaller sections in between these barriers. This barrier effect is reflected in the modeling approach by using a permanent ridge-tillage sequence, which functions as a ridge-furrow system and influences runoff and the transport of sediment from interrill to rill areas. As the undisturbed area between single ridges is relatively small (distance between stone lines is 5-10 m), the upslope area is limited, and simulated runoff and soil loss are accordingly low. This effect could also explain the low amplitude of variations between individual hillslope positions.

Simulated soil deposition corresponds well with the magnitude of eroded sediment transported downslope and increases in accordance with the observed soil loss. Especially at the footslope position (OFE 5), high deposition rates of 22.3-40.5 t ha^{-1} y^{-1} for conservation methods and 94.6-212 t ha^{-1} y^{-1} for conventional practices are observed, reflecting the effect of abruptly decreased slope gradients and diminished sediment transport capacities, which lead to high redeposition rates of upslope sediment. In contrast to all other practices, soil deposition does not occur at all when stone lines are integrated, which indicates that the WEPP model might not reflect the sediment trapping factor for ridges as erosion barriers.

In order to identify which of the tillage methods is most suitable to minimize soil loss on erosion-prone hillslope positions, changes in simulated soil loss were calculated for all land management options based on a control run and compared with each other for individual hillslope positions (Table 3.9).

Table 3.9 Soil erosion/deposition simulated by WEPP model for transect 1 at Dano, Burkina Faso

WEPP-simulation		HT ($t\ ha^{-1}\ yr^{-1}$) Control run	**Fractional changes for each land management option compared with simulated actual soil loss** RA (%)	MT (%)	B-75 (%)	B-50 (%)	CP (%)	SL-5m HT (%)	SL-10m HT (%)	SL-15m HT (%)	SL-5m MT (%)	SL-5m-10m MT (%)	SL-5m MT RA (%)
OFE 1	E	-4.7.	-7.9	-11.7	201	62.9	326.7	-11.3	1.1	33.0	-11.3	-11.3	-13.6
	D	-	-	-	-	-	-	-	-	-	-	-	-
OFE 2	E	-27.6	-27.2	-38.8	279.6	115.2	626.8	-87.0	-85.4	-80.9	-87.0	-87.0	-87.8
	D	-	-	-	-	-	-	-	-	-	-	-	-
OFE 3	E	-90.5	-18.6	-23.1	292.1	130.5	342.2	-95.2	-94.8	-93.6	-95.2	-95.2	-95.4
	D	-	-	-	-	-	-	-	-	-	-	-	-
OFE 4	E	-101.9	-22.4	-23.0	313.4	121.1	221.7	-95.6	-95.3	-94.0	-95.6	-95.6	-95.7
	D	-	-	-	-	-	-	-	-	-	-	-	-
OFE 5	E	-15.0	-28.3	-41.1	293.5	125.7	106.5	-73.3	-71.2	-65.7	-71.2	-73.3	-73.3
	D	29.2	-23.5	39.0	569.4	224.7	626.7						
OFE 6	E	-8.5	-22.0	-37.2	252.1	71.8	741.3	-51.7	-49.6	-42.6	-49.6	-51.7	-51.7
	D	5.7	-47.0	-45.2	361.8	193.8	114.1	-47.0	-	-	-	-	-
Net erosion ($t\ ha^{-1}\ yr^{-1}$)		-35.2	-27.4	-25.0	-125.3	-72.1	-113.5	-4.1	-4.4	-5.4	-4.2	-4.1	-4.1
Fractional changes (%)		-	-22.0	-29.0	256.4	105.2	222.9	-88.3	-87.4	-84.6	-88.0	-88.3	-88.5
Sediment delivery ratio (%)		90.8	90.4	87.4	82.3	84.9	71.6	99.0	99.2	99.4	99.1	99.0	99.1

E: Erosion; D: Deposition; HT: Handhoe tillage; RA: Residue addition; MT: Minimum tillage; B-75: Burning-75 % residue lost; B-50: Burning-50 % residue lost; CP: Chisel plow tillage; SL: Stone lines; 5m/10m: 5m/10 m spacing between stone lines; positive values indicate soil gain, negative values soil loss. Grey color: Control run.

Simulated soil loss under handhoe tillage was chosen as the base value for the control run, because this practice is known to be common on the hillslope. Fractional changes show that net erosion rates calculated for the entire hillslope can be reduced by 22 %, 29 % and more than 85 % under residue addition, minimum tillage and stone line application, respectively. Burning and high-tech tillage will increase erosion rates by 100-250 %. At specific hillslope positions, the effect of land management options is even more distinct. A reduction in soil loss of approximately 40 % can be achieved under minimum tillage at shoulder and footslope position (OFE 2 and OFE 5), and 95 % under stone lines at backslope position (OFE 3), whereas chisel-plough tillage practices would lead to six to seven times higher rates if used at the shoulder and toeslope/valley positions (OFE 2 and OFE 6).

Land management options at the hillslope in Wahable illustrate that simulated soil loss rates are up to ten times lower than in Dano, and the amplitude of variations between both individual land management practices and specific hillslope positions is smaller (Figure 3.17). Whereas conservation techniques, such as minimum tillage and residue management, at backslope position (OFE 3) show maximum soil loss rates of 3.0 and 5.7 t ha^{-1} y^{-1}, conventional practices, such as ox-plough tillage, burning and chisel-plough tillage, can lead to up to 10.6, 29.4 and 31.6 t ha^{-1} y^{-1}, respectively. Additionally, the effect of tillage along contour lines is simulated. Here also, a ridge-tillage sequence was applied with a spacing between contour ridges of 0.8 m and a ridge height of 0.1 m, which allows overtopping of runoff in case of heavy rainstorms. Soil erosion rates range from 2.8 t ha^{-1} y^{-1} to 4.6 t ha^{-1} y^{-1} and again show no significant variations between individual slope positions. Deposition rates of 1.7-3.9 t ha^{-1} y^{-1} for conservation methods and 5.9-18.2 t ha^{-1} y^{-1} for conventional practices can be observed at the footslope and toeslope/valley position (OFE 4 and OFE 5).

Fractional changes were based on control values for ox-plough tillage, which is the actual land management practice at Wahable and also represents a common and potential future land management practice for farmers (Table 3.10). Net erosion rates for the entire hillslope can be reduced by 36-65 % under minimum tillage, residue addition or a combination of both. Contour tillage leads only to an average reduction of 4 %, but can reach 36 % if supplemented by residue addition.

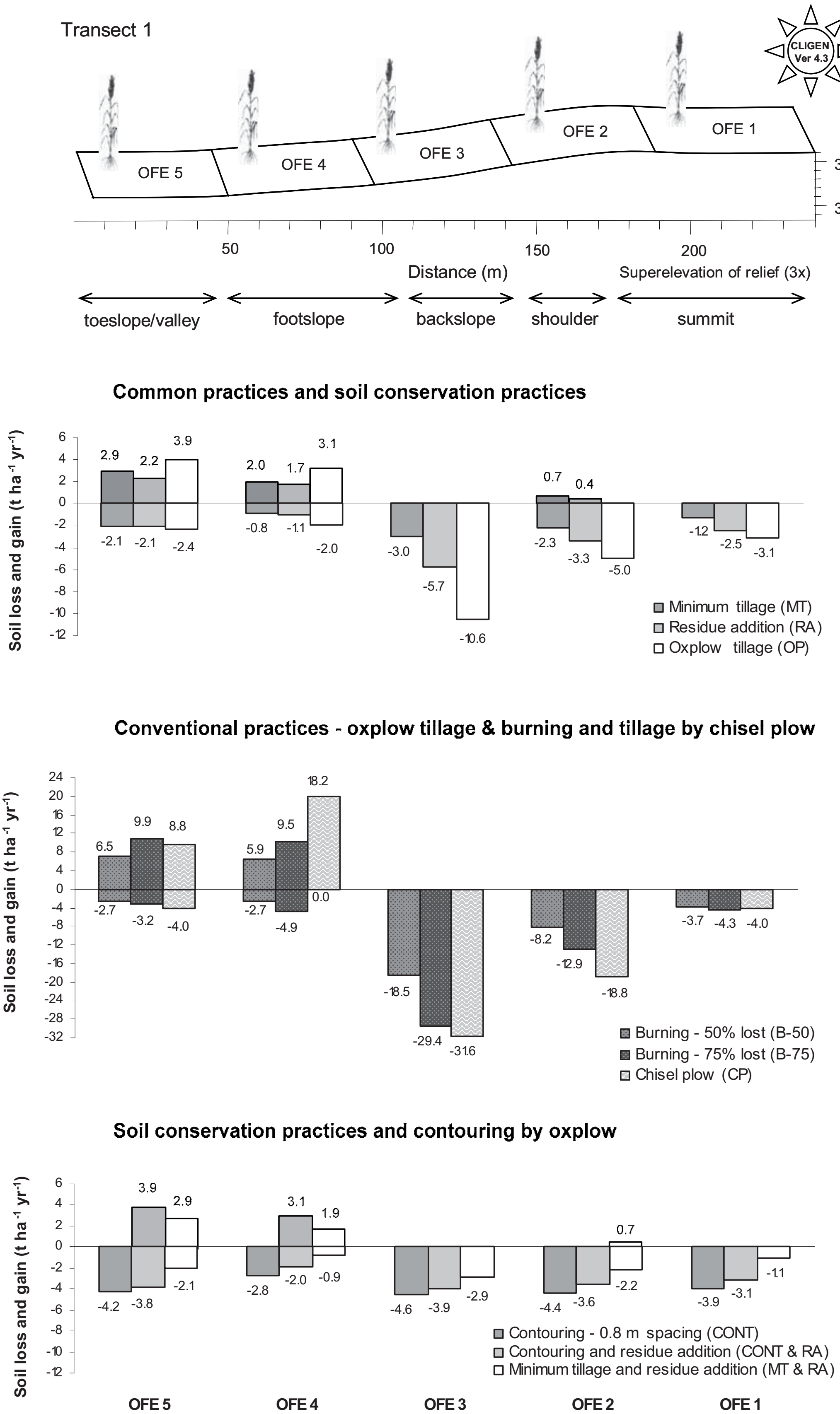


Figure 3.17 Effect of land management options on simulated soil loss at Wahable

Table 3.10 Soil erosion/deposition simulated by WEPP model for transect 1 at Wahable, Burkina Faso

WEPP-simulation		**Fractional changes for each land management option compared to simulated soil loss (OP)**								
		OP ($t\ ha^{-1}\ yr^{-1}$) Control run	RA (%)	MT (%)	B-75 (%)	B-50 (%)	CP (%)	CONT (%)	CONT RA (%)	MT RA (%)
OFE 1	E	-3.1	-20.7	-61.2	38.2	18.5	28.7	24.2	0.00	-64.7
	D	-	-	-	-	-	-	-	-	-
OFE 2	E	-5.0	-33.7	-55.2	155.4	62.5	272.2	-13.1	-28.2	-56.4
	D	-	-	-	-	-	-	-	-	-
OFE 3	E	-10.6	-45.9	-71.8	177.3	74.7	197.6	-56.7	-63.2	-73.0
	D	-	-	-	-	-	-	-	-	-
OFE 4	E	-2.0	-45.9	-58.2	148.5	36.7	-	41.8	0.00	-56.6
	D	3.1	-44.5	-37.1	205.5	89.4	486.5	-	-	-39.4
OFE 5	E	-2.4	-11.9	-12.3	37.9	13.2	70.2	80.4	62.6	-12.8
	D	3.9	-44.5	-26.7	150.9	65.1	123.9	-	-	-27.0
Net erosion rate ($t\ ha^{-1}\ yr^{-1}$)		-4.2	-2.7	-1.5	-9.2	-6.2	-8.1	-4.1	-3.3	-1.5
Fractional changes (%)		-	-35.8	-63.5	118.0	46.6	90.5	-3.7	-21.0	-65.0
Sediment delivery ratio (%)		92.8	94.1	87.9	86.1	87.7	66.6	100.0	100.0	87.8

E: Erosion; D: Deposition; OP: Ox-plough tillage; RA: Residue addition; MT: Minimum tillage; B-75: Burning-75 % residue lost; B-50: Burning-50 % residue lost; CP: Chisel plow tillage; CONT: Contour farming by ox-plough; positive values indicate soil gain, negative values soil loss.Grey color: Control run.

When looking at individual hillslope sections, however, the effect of contour farming is higher, resulting in approximately 60 % lower soil loss at the backslope position (OFE 3), which is especially prone to erosion. In contrast, burning and high-tech tillage would increase average erosion rates by approximately 50-100 %, and could provoke three times higher erosion rates at the most susceptible backslope positions. Whereas erosion simulations have shown the effect of different land management options on soil loss, further investigations should focus on the advantages and limitations and on the feasibility to apply these options in Burkina Faso (Table 3.11).

For simulation purposes, the choice of land management options had to be selective, and was chosen exemplarily from a wide range of available agronomical (e.g., crop rotation, fallowing, intercropping, no-till), biological (e.g., stone lines, grass strips, hedges, mulching and manure application) and mechanical (e.g., contour farming, terracing, earth bunds) practices.

Traditional farming practices, such handhoe tillage or shallow tillage by ox-plough, are not sufficient to minimize the risk of physical soil degradation and to prevent the nutrient-rich surface layer from being eroded (Kiepe et al., 2001b; Pieri, 1989). On soils that show severe sealing, crusting or hardsetting characteristics, a mechanical loosening and breaking up of the soil structure is often proposed as necessary to improve permeability and water infiltration capacity and to provide better conditions for an extended root system (Nicou et al., 1993; Valentin, 1993). In such cases, also minimum tillage is not recommended as an adequate option, because it will not improve the soil structure or replenish the poor nutrient status of the soil. Considering the practicability of these options for farmers, both handhoe and minimum tillage present the most feasible options. First of all, they have been traditionally practices for many years and secondly, they demand neither high input costs nor technical investments. Nevertheless, tillage by handhoe might be very labor intensive, especially on degraded and hard-cover soils. Tillage by animal traction, however, requires high initial investments for tillage implements and the oxen and some, although relatively low, maintenance costs. Advantages are the quick, mechanical preparation of fields for sowing and the low labor demand. Where oxen and tillage implements are available or can be borrowed, animal traction has become a widespread option and is easily adapted by farmers (Mazzucato and Niemeijer, 2000b; Veihe, 2000).

Table 3.11 Evaluation of the application of selected land management options in Burkina Faso

Measure	Effect on soil loss: Runoff plots	Effect on soil loss: WEPP simulation	Advantages	Limitations	Feasibility for farmers
Handhoe tillage (HT)	-	Control run	Low soil surface disturbance, low costs, low inputs	No improvement of soil structure, no amelioration of nutrient status, labor intensive	+
Ox-plough tillage (OT)	-	Control run	Fast, mechanical technique on larger fields, low labor demand	High soil compaction, decreased infiltration, high costs for ox-plough	- - -
Minimum tillage (MT)	-	-30 to -70 %	Minimum soil surface disturbance by planting holes, low costs/ inputs	No improvement of soil structure, no amelioration of nutrient status	+
Mulching/ residue addition (RA)	-40 to -95 % [c]	-20 to -45 %	Reduced splash effect due to denser soil cover; increased soil organic matter, infiltration and biological activities	Availability of straw, mulch, and/or residue; increase in insects, pests and parasites, labor intensive, know-how required	-
Stone Lines (SL)	-15 % [a] -45 % [b]	-85 to -95 %	Reduced runoff and soil loss, moisture retention, increased infiltration; prevention of rill and/or gully formation	Availability of stones; high investment costs, transport facilities and costs; labor intensive; know-how required	- -
Contour farming (CONT)	-30 % [c]	-4 to -55 %	Reduced runoff and soil loss, moisture retention; prevention of rill and/or gully formation	Not appropriate for steep slopes	-
Burning (B)	+300 % [c]	+100 to +250 %	Quick clearing of land; natural weed and pest control; rapid nutrient release	Long-term loss of nutrients, loss of soil fertility if annually practiced without adequate fallow periods	+

[a] Lamacher and Serpatie, 1990; [b] Zougmoré et al., 2004a; [c] Roose, 1994
(feasibility for farmers: + good ; - medium to low ; -- low ; --- very low)

In the case of biological practices, mulching (referred to as residue addition in simulations) is one of the most effective methods to restore soil organic matter in the humus layer by covering the soil surface with residue or straw (Lal, 2000; Mando and Stroosnijder, 1999; Kiepe et al., 2001b). Especially during the dry season, an adequate soil coverage is necessary to minimize the risk of soil erosion. Already thirty years ago, Roose (1976) concluded that artificial mulch was very effective in reducing soil loss by 40-95 %, and that even straw mulch of a few centimeter thickness had the same effect as a secondary forest of 30-m height in absorbing kinetic rainfall energy. Soil loss simulations in this study confirm this positive effect of residues and show that average soil loss could be reduced by 20 to 35 % and up to 45 % at erosion-prone hillslope positions. Furthermore, mulching can ameliorate the soil fauna by attracting termites and other soil organisms, which will lead to increased biological activity and enhanced mineralization of soil material (Mando, 1997). Biological activities will also improve the physical structure of the soil by bioturbation, which affects soil porosity and bulk density and facilitates water infiltration and hydraulic conductivity (Mando et al., 1996). However, the bottleneck of mulching is that it is difficult to provide sufficient quantities of vegetal residue to cover the fields adequately and to replenish or even maintain the nutrient status of the soil (Giller et al., 2009). A thick mulch of 20 to 25 t ha^{-1} would be optimal for reducing the splash effect, retaining soil moisture during the dry season and restoring nutrients. But even a light mulch of 2 to 6 t ha^{-1} can have already a significant impact and can reduce the erosion risk by 80 % even if only 50 % of the soil is covered (Roose, 1994). However, if residues are insufficient for a light mulch, supplementary residue production or the purchase of artificial mulch is for most farmers either not practicable or too costly. Unfortunately, some farmers are also concerned about the increase in biological activity, as they assume that termites, soil organisms and insects might be a pest risk and hence affect yields (Veihe, 2000).

Another promising soil conservation technique for ameliorating the soil structure, and at the same time for minimizing runoff and soil loss, is the construction of stone lines (Mietton, 1986; Roose, 1994; Zougmoré et al., 2004a). In northern Burkina Faso, Lamacheré and Serpantie (1990) found that stone lines could increase the infiltration capacity of the soil by 15 % on sandy to sandy-clay soils. Zougmoré et al. (2004a) also experienced that runoff was reduced by 45 % when stone lines were constructed. This corresponds with the WEPP model simulations, where stone lines showed an even higher impact of 95 % soil loss reduction on steeper hillslopes positions. However, the construction of stone lines depends first of all on the availability of stones, which can be the limiting cost factor for farmers if stones need to be transported (De Graaff and Spaan, 2002; Veihe, 2000; Zougmoré et al., 2004a).

Furthermore, the initial set up and construction is very labor intensive. A certain know-how is required to maintain the functionality of the stone lines and to maximize their potential as runoff and sediment barriers, otherwise they can even provoke lateral rills and/or gully formation (Roose, 1994). Nevertheless, if the initial implementation and transport costs are covered by external resources (e.g., governmental programs, NGOs or projects), if continuous support and maintenance is ensured and, last but not least, if farmers participation and initiative is given, they can be a very effective option to counteract soil erosion (Batterbury, 1994; Reji et al., 2005; Reji and Smaling, 2007).

Contour farming presents another feasible option and a commonly used method to improve water infiltration capacity. Roose (1994) found that contour ridging could reduce erosion by 30 % on experimental plots with slope gradients of 1-8 %. In this study, simulated soil loss could be reduced by 4 % in average for the entire hillslope, and by 55 % at specific hillslope positions. But the efficiency of contouring decreases as slope gradients increase. On very steep slopes, contouring should be accompanied by grass strips to avoid the risk of lateral rill or even gully formation. Also, a combination with residue addition, mulching or manure application is often proposed to increase soil organic matter (Kiepe, 2001a).

Burning is mainly used for clearing of land, for natural weed and pest control, and for quick access to stored nutrients locked up in the vegetation. Nevertheless, nutrients released from ashes are short-term fertility solutions, as they cannot sustain an even low agricultural production for more than five years in the savannah zone (Pieri, 1989). In such slash-and-burn systems, long fallow periods of several years are necessary to re-establish the initial vegetation cover and to restore nutrients. However, if adequate fallow periods are adhered to, they might be sustainable and efficient land management systems (Valentin et al., 2004; Goudie, 1993). Burning performed up to one month after the last rainfall passes rapidly over the fields and affects mainly withered plant material, dead biomass, young seeds, soil organisms and insects, and is therefore often used for weed and pest control. Depending on extent and time, these early burnings can affect up to 50 % of the vegetation cover, and might cause three times higher soil loss compared to erosion plots with sufficient protective vegetation cover (Roose, 1994). Similar high soil loss rates of 100-300 % were also simulated by the WEPP model.

The former simulation results and the above reflection about appropriate land management options indicate that soil loss varies highly along the hillslope, but can be reduced significantly by applying adequate soil and water conservation methods on erosion-prone hillslope positions. Furthermore, considering the high erosion risk and the low nutrient status of the soils in Burkina Faso, soil conservation techniques should have both a physical-mechanical function to control runoff and soil loss and a biological-agronomical impact to restore soil nutrients and improve soil fertility. A combination of both measures would be the most appropriate option to meet the concept of sustainable land management.

For future application, an integration or coupling of soil erosion and soil nutrient models would be a promising step forward to compensate nutrient losses due to soil erosion. Such an integrative model should be spatially explicit to simulate detachment, transport and deposition of nutrient-rich sediment downslope and to identify nutrient deficiencies.

3.5.9 Comparison between simulated soil erosion rates by WEPP model and ^{137}Cs measurements

Simulated WEPP model results are compared with ^{137}Cs measurements by plotting soil erosion/deposition estimates derived from the ^{137}Cs method against those simulated by the WEPP model. Both the ^{137}Cs mass-balance model 3 and the hillslope version of the WEPP model calculate soil loss/gain along individual transect lines, which allows a direct comparison of erosion/deposition rates for exactly the same position at the hillslope. The scatter diagram illustrates the agreement between both methods when ^{137}Cs estimates are assumed to be correct (Figure 3.18). The diagonal line represents the perfect 1:1 relationship. The closer the simulated results to the 1:1 line, the more accurate is the WEPP model. For both study sites, the scatter diagrams show that the data points of the model diverge highly from the diagonal line, indicating a week relationship between both methods. This observation is also confirmed by the high Root Mean Square Error (RMSE) of 49.8 for Dano and 19.4 for Wahable.

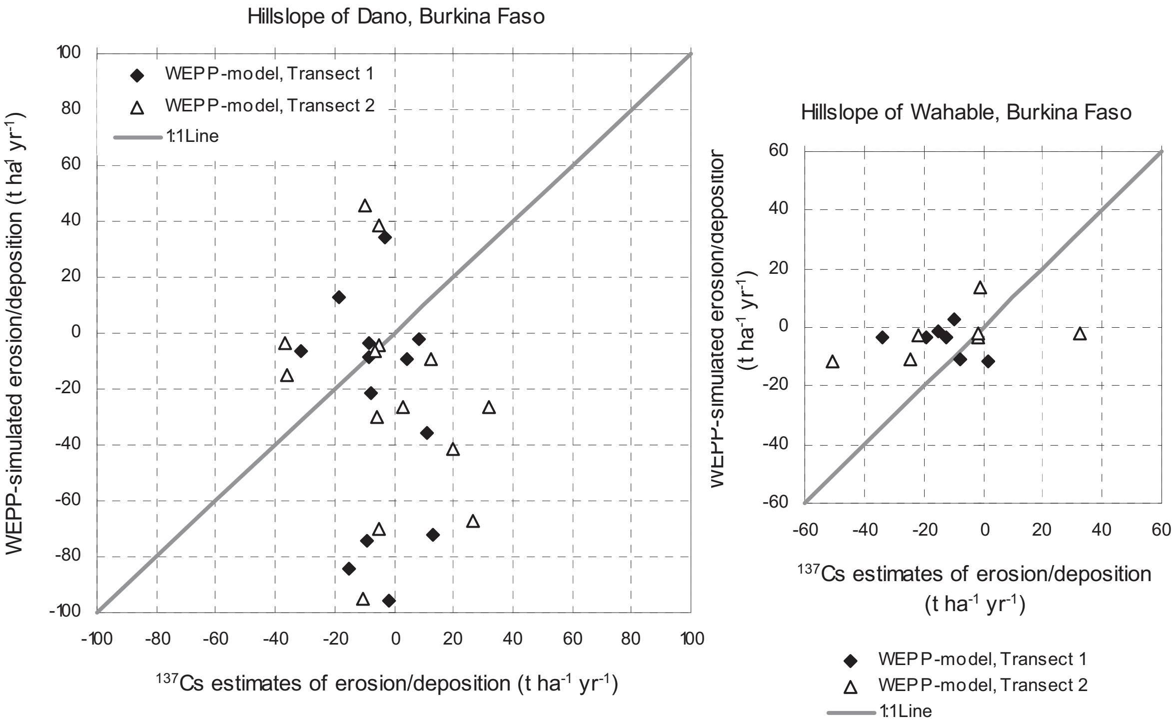


Figure 3.18 Comparison between ^{137}Cs estimates and WEPP model simulated erosion and deposition rates at both study sites in Burkina Faso

The high RMSE and the low agreement between both methods indicate that either one or both methods might be inaccurate. This might be due to shortcomings of each approach, such as internal model limitations or measurement errors, as well as to uncertainties in measuring environmental variability and complexity. Some of the shortcomings of both approaches have been already discussed (sections 3.5.5, 3.5.6 and 3.5.7) and will be summarized here only briefly.

The hillslope version of the WEPP model showed that parallel soil profiles along a hillslope might produce highly different erosion rates due to small changes in soil properties, such as soil texture (section 3.5.7). The model clearly overestimated soil loss for the clayey soils in Dano, which might be related to the relevance of clay content for pedotransfer functions (e.g., to calculate bulk density, soil water content, soil porosity, effective hydraulic conductivity and rill erodibility; Alberts et al., 1995). The high erosion rates for Dano and hence the large differences between ^{137}Cs measurements and WEPP model simulations might be also caused by the model's neglect of vertical pedogenesis and illuvation or weathering processes. The model's low sensitivity to soil input parameters beyond a depth of 20 cm means that changes in soil texture, organic matter, CEC or rock fragments at lower soil horizons cannot be accounted for. This might be particularly relevant for soil profiles characterized by illuvation horizons, plinthite layers or soft bedrock, and for simulations over multiple decades. All of these criteria apply to the study site in Dano, where iron-rich plinthite layers are found at soil depths of 22 cm (Profile R3, Figure 3.4a), which greatly restrict water permeability. If these soil horizon characteristics in the lower layers are not considered in mid- to long-term simulations, processes of vertical soil-water movement, hydraulic conductivity and infiltration capacity might be oversimplified, and the effects not sufficiently reflected in soil loss predictions.

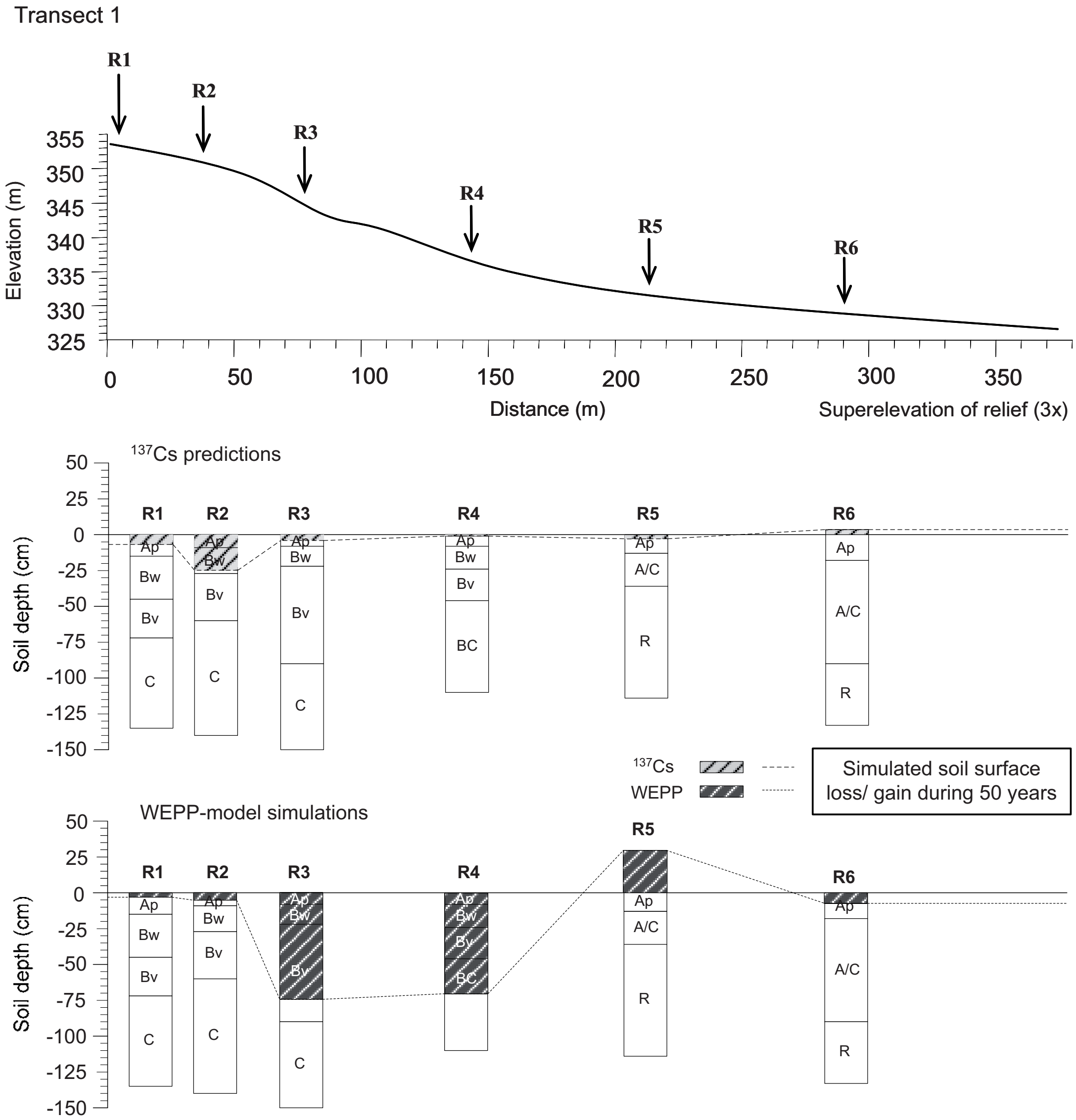
The ^{137}Cs approach considers these shortcomings by measuring the amount of ^{137}Cs in the upper 30 cm of the soil profile, where the ^{137}Cs content serves as an indirect indicator for mid-term soil loss. The ^{137}Cs approach represents an in-situ measurement of soil status quo, and is a retrospective evaluation of soil erosion during the last 50 years. This implies that the approach deals with a high input uncertainty in regard to environmental variability. Spatial variability is reflected by fixed parameter values, e.g., bulk density, plough depth, proportion factor and tillage constant. Temporal variability is reduced to one specific point in time determined by the collection of soil samples for model calculations. Therefore, additional historical land-use information is needed, which, however, is often based solely on farmers' memories. Similar concerns apply to the reference sample as the most important parameter for transferring ^{137}Cs inventories into soil redistribution rates. Further shortcomings are related to parameter uncertainties of the conversion models, such as a fixed relaxation depth or an assumed static particle size correction factor accounting for the absorption of ^{137}Cs by soil particles. Variations in the latter parameter could lead to changes of 10-20 % in soil loss rates.

Although both approaches are based on 50-year simulation runs and require actual parameter values for soil, land-use and land management, they follow a different time line (retrospective versus prospective focus), which restricts the comparability of simulated results. Whereas the ^{137}Cs approach represents an ex-post evaluation and considers soil erosion in a retrospective view (based on initial radionuclide fallout in 1954 until the date of sample collection in 2005/06), the WEPP model approach can be seen as an ex-ante simulation for prospective future soil erosion scenarios (starting from the sample collection in 2005/06 until an optional future date in 2055). Additionally, both approaches assume water as the dominant agent for soil erosion and neglect the impact of wind erosion as an additional major agent for soil detachment, transport and moreover accumulation in semi-arid environments (Chappell, 1999).

However, even though a close agreement between WEPP model simulations and ^{137}Cs estimates could allow some degree of validation, it does not confirm explicitly the validity of the internal functioning of the model and finally of the predicted soil redistribution rates. Furthermore, Walling and He (1998) pointed out that a "close correspondence between observed and predicted outputs could, for example, still be obtained in situations where the erosion and deposition rates predicted by the model differed substantially from the actual values" (Walling and He, 1998, p. 267).

Therefore, another question should be asked: What kind of implications do these soil erosion predictions involve and what can be concluded from simulated soil redistribution rates for soils in Burkina Faso? In this context, the former interpretation of soil profiles provides an option to illustrate how much of the soil profile will be removed if soil erosion continues over a time period of 50 years (Figure 3.19 and 3.20). Hence, predicted annual soil loss rates were converted into the magnitude of soil-surface lowering by multiplying mid-term soil loss rates by bulk density values for the A-horizon. The extent of soil-surface lowering expressed as soil loss depth from profiles presents a more qualitative measure of the severity of soil erosion and indicates the loss of nutrient-rich topsoil. Surface lowering estimated by ^{137}Cs calculations would lead to 25 cm at the shoulder position (R2). The WEPP model simulations, in contrast, show a 70-75 cm soil-profile loss at the most affected backslope positions (R3 and R4). Deposition-related surface increase is less pronounced in ^{137}Cs calculations and leads to 3.5 cm soil profile augmentation at toeslope/valley position (R6), whereas WEPP model simulations show an increase of 30 cm at footslope position (R5).

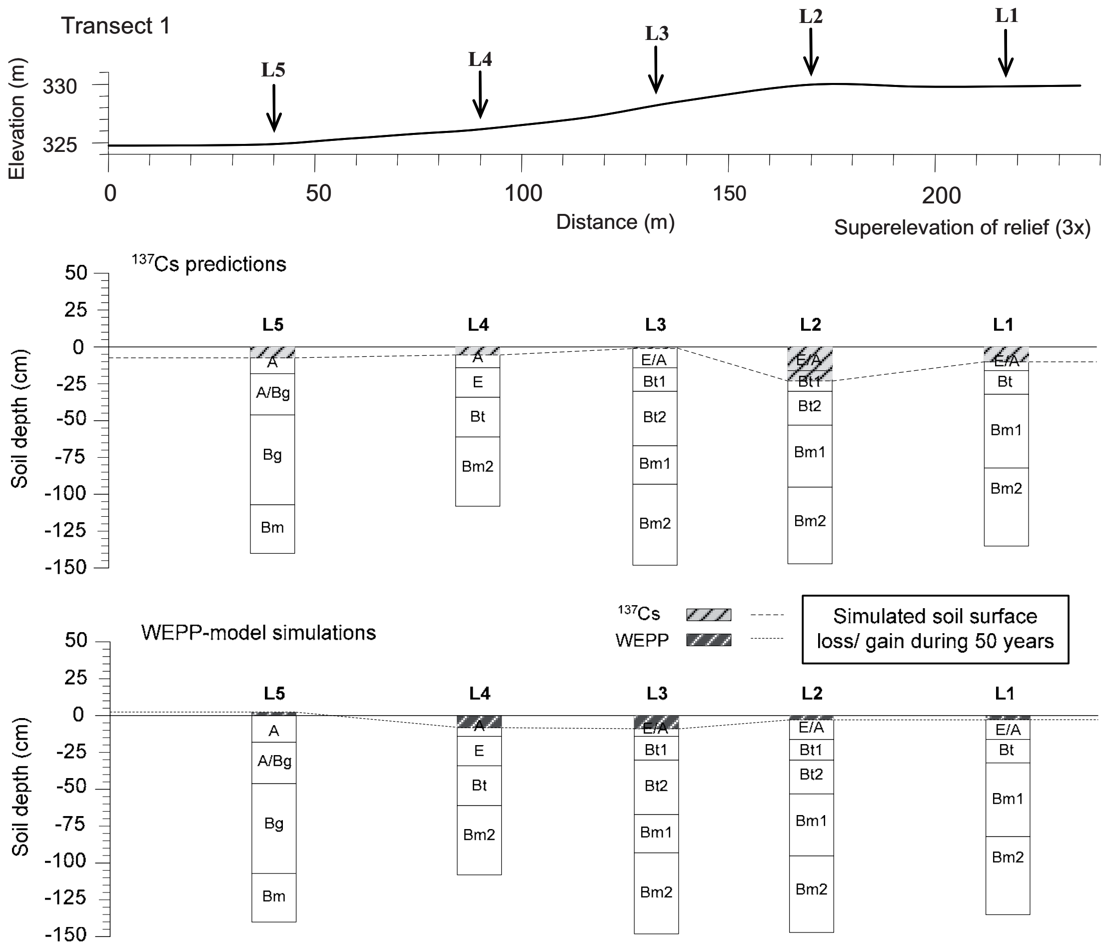
Similar results were obtained for the site in Wahable, where the most severe soil profile loss is estimated to be 23 cm at shoulder position (L2) by ^{137}Cs calculations and 8.6 cm at backslope position (L3) by WEPP model predictions (Figure 3.20). Surface gain by sediment deposition is solely predicted by the WEPP model with 2 cm sediment yield at toeslope/valley position.



Abbreviations for master and transitional horizons (Schoeneberger et al., 2002)

A: Mineral soil, formed at surface; accumulation of humified organic matter but dominated by mineral matter; cultivation properties. A/B or A/C: Discrete, intermingled bodies of two horizons: A and B or C material, majority of layer is A horizon material. B: Mineral soil, typically formed below O,A, or E (typical features e.g. illuvial accumulation, gleying). B/C: Dominantly B horizon characteristics but also has some recognizable characteristics of the C horizon. C: Mineral soil, soft bedrock; layer little affected by pedogenesis; may or may not be related to parent material of the solum. R: Hard bedrock (continuous, coherent strongly to indurated cemented. p: Tillage or other disturbance of surface layer (e.g. pasture, plow). v: Plinthite (high Fe, low OM, reddish contents; firm to very firm moist consistence; irreversible hardening with repeated wetting and drying). w: Minimal illuviation accumulation.

Figure 3.19 Soil-surface loss and gain based on 50-year simulation by WEPP model and ^{137}Cs measurements for transect 1 at Dano, Burkina Faso



Abbreviations for master and transitional horizons (Schoeneberger et al., 2002)

A: Mineral soil, formed at surface; accumulation of humified organic matter but dominated by mineral matter; cultivation properties. A/B or A/C: Discrete, intermingled bodies of two horizons: A and B or C material, majority of layer is A horizon material. B: Mineral soil, typically formed below O,A, or E (typical features e.g. illuvial accumulation, gleying). E: Mineral soil, loss of clay, iron, aluminum and/or organic matter leaving a net concentration of sand and silt; lighter color and coarser texture. E/A: Discrete, intermingled bodies of two horizons: E and A material, majority of layer is E horizon material. g: Strong gley, typically = 2 chroma; may have other redoximorphic (RMF) features. m: Strong pedogenic cementation or induration (>90% cemented, even if fractured); physically root restrictive. t: Illuvial accumulation of silicate clays (clayskins, lamelae or clay bridging).

Figure 3.20 Soil-surface loss and gain based on 50-year simulation by WEPP model and ^{137}Cs measurements for transect 1 at Wahable, Burkina Faso

The magnitude of soil-surface lowering and hence soil profile reduction shows that in a short time period the nutrient-rich A-horizon will be partially or completely removed at all erosion-affected hillslope positions. Most of the primary nutrients essential for crop growth are stored in the upper soil horizon, and its loss would seriously affect soil fertility and hence plant growth. As the analysis of soil horizon properties has already shown that the nutrient status of the soils is highly deficient especially for available P and K (section 3.5.2), a further deterioration due to soil erosion would aggravate these nutrient deficiencies.

Another problem of soil profile lowering will be the exposition of plinthite-rich subsurface layers. The pedo-geomorphic description of soil development confirmed high concentrations of manganese and iron oxide at lower soil depth in Wahable (section 3.5.1), which could lead to irreversible hardening and hence the formation of laterite crusts if these lower soil horizons would be exposed to the air. According to 50-year simulation runs, soil profiles at erosion-prone hillslope sections, such as shoulder (R2) and backslope position (R3 and R4), would be affected by this laterization, and once laterite crusts have been formed, the soils would not longer be suitable for agriculture.

The bottleneck of these calculations is certainly the neglect of pedogenesis and weathering processes in the predictions. Dynamic interactions and feedback mechanisms are not considered by the models, and the abrupt change in soil profile characteristics is not reflected in adjusted soil loss rates. If, for example, the loss of the complete overlying A-horizon and thus the exposure of a more cemented soil horizon (e.g., Bw) is predicted, models should account for the new, corresponding soil properties and incorporate them in their internal structure. The integration of such a process-response interaction would allow a more dynamic modeling framework, account for negative or positive feedback mechanisms, and hence a better adjusted calculation of soil erosion rates. In the present study, soil loss rates would have been highly reduced and soil-surface lowering would have been much less.

However, the purpose of simulating soil-surface lowering is neither another comparison of the prediction capability of the models nor a further evaluation of limitations and shortcomings of both approaches, but more a suitable and practical evaluation of the impact of soil erosion. Although great differences in the magnitude of soil-surface lowering exist, both predictions indicate that the A-horizon is highly endangered if no prevention strategies, soil and water conservation methods, or adequate land management practices are implemented. Furthermore, at both study sites, soil loss appears also in form of sheet erosion, which is often unnoticed by farmers due to the absence of visible signs (e.g., erosion rills) and explains why no African dialect has a specific word for this process (Roose et al., 1994) although its impact is high.

3.6 Conclusion

The assessment of soil erosion at hillslope scale has shown that soil loss and deposition vary highly in dependence with hillslope position and cannot be captured by single numbers nor by averaged erosion or deposition rates for an entire hillslope. Compared to averaged erosion rates, net erosion rates can be up to forty times higher on steeper hillslope sections (e.g., Dano) and still up to ten times higher on almost flat hillslopes (e.g., Wahable). Furthermore, soil loss and deposition rates differ due to the approach used and absolute values should be considered with care. This was also indicated by the low correspondence between ^{137}Cs results and WEPP model simulations for the same study sites. The significant differences in predicted results could be caused by their different methodological framework and time focus. Whereas the ^{137}Cs method calculates soil erosion in a retrospective, ex-post view by considering the amount of ^{137}Cs removed during the last approximately 50 years, the WEPP model simulates future soil erosion rates in 50 years from a prospective, ex-ante view based on the current status of the soil. Therefore, an inverse modeling approach should be considered in further erosion scenarios to simulate soil loss for the same time period and hence to allow a better comparison with retrospective soil loss measurements.

At both sites, the catenary concept served as a suitable framework to interpret the systematic sequence of soils and to consider the interrelation between slope geometry and variations in physical and hydrological properties. The spatial variability of inherent soil characteristics (e.g., eluviated horizons, leaching, hydromorphic features) could be related to distinct hillslope positions as well as variations in soil nutrients, although the reciprocal relationship for the latter was less defined due to high nutrient deficiencies and interference with land management practices. The clear differentiation of the hillslope into distinct hillslope positions, confirmed firstly the validity of the catena concept for soils in southwestern Burkina Faso and presented secondly a tool to include the variability of soils and hence pedo-geomorphic process units in soil erosion simulations.

The physically-based WEPP model showed that the catena concept could be integrated into the modelling approach by dividing the hillslope into several distinct hillslope positions (such as summit, shoulder, backslope, footslope and toeslope/valley) so that simulations could better account for the demarcation of erosion and sedimentation zones. In this regard, field units should not be considered as closed entities, moreover as parts of a larger, complex hillslope system where erosion and sedimentation processes are not limited to field boundaries but interact in a interconnected process-responds system over the entire hillslope. Within this small-scale system, distinct hillslope position will represent spatially explicit erosion and deposition zones reflecting individual pedo-geomorphological and hydrological processes. The identification of these spatially explicit zones can be used in modelling approaches in order to select the most appropriate land management practice for the individual hillslope positions. The WEPP model simulations have shown that at steeper hillslope positions (such as shoulder and backslope position), the application of stone lines, minimum tillage, contour farming and residue management could reduce soil loss by maximal 95 %, 70 %, 55 % and 45 % respectively, whereas burning and tillage by chisel plow could aggravate soil loss by maximal 250 % and 600 % compared to the actual land management practices. This indicates that especially at erosion-prone hillslope positions, adequate land management options could highly reduce the risk of soil erosion. While, however, erosion simulations could facilitate the selection of most-effective land management techniques in terms of soil loss reduction, the feasibility for farmers to apply these techniques in Burkina Faso in terms of socio-economical constrains (e.g., cost, labour, implementation) should be also taken into consideration because it can restrict highly the actual feasibility of conservation methods. The choice of potential options in land management practices depends equally on the effectiveness of the method as on the practicability for farmers. Therefore, future research should focus on both, ecological factors as well as socio-economical aspects in order to develop decision-support systems on which higher level decisions for more appropriate and sustainable land management can be based.

For farmers, the most evident consequences of soil erosion are on-site effects occurring directly on their field. These on-site effects are related to the loss of nutrients-rich topsoils and/or soil deterioration through the formation of laterite crusts, whereof the latter is still not considered by any soil erosion model. The consideration of nutrient management would be especially valuable in such environments as Burkina Faso as soils are highly deficient in primary nutrients while they present at the same time the most important resource to assure crop yield and to provide finally food security for an increasing agricultural society. Future soil erosion models should therefore couple, link or integrate plant nutrient models with soil erosion models in order to quantify nutrient loss as a consequence of soil erosion and to account for better options and land management techniques to preserve and restore soil nutrients. Also, if soil erosion models provide simulations for different land management options, scenarios should reflect practicable soil conservation methods for West Africa (e.g., contour farming, stone rows) and they should integrate possible nutrient management practices (e.g., mulching, residue, fertilization) to simulate possible pathways for sustainable agriculture and erosion risk alleviation.

4 SOIL EROSION AND RESERVOIR SILTATION AT CATCHMENT SCALE

4.1 Introduction

Soil erosion assessment at catchment scale allows accounting for both on-site and off-site impacts of soil erosion. Whereas on-site impacts can be adequately identified at hillslope scale as they include soil loss processes occurring directly on farmers fields, off-site impacts require a view at a larger scale as they comprise additional processes related to sediment mobilization, sediment transport and sediment delivery beyond the limited boundaries of a hillslope (Kirkby, 1996; Rickson, 2006; Sivapalan, 2003).

The catchment scale is therefore widely seen as the most appropriate scale to combine topological, pedological and land-cover properties on-site with sediment fluxes and runoff routing characteristics between individual fields, hillslopes and landscape units (Grayson and Blöschl, 2000; Zhang et al., 2004). By analyzing these interactive dynamics, soil erosion becomes a complex sediment delivery process, in which topsoil is eroded in source areas, routed by flow networks downslope and finally accumulated in intermediate sinks or in long(er)-term storages within or at the outlet of the catchment. In order to identify such sink and source areas, sediment budgeting is often used as a holistic approach to calculate the mass balance of catchments (Slaymaker, 2006; Smith and Dragovich, 2008; Wilkinson et al., 2009; Walling and Collins, 2008). Sediment yield, which refers to the amount of sediment exported to the outlet of the catchment (Morris and Fan, 1997), presents the ultimate parameter of the sediment budget approach. Although sediment yield might not provide adequate information about the variability of soil erosion within the catchment, it can be used as a valuable indicator to determine the quantity of sediment delivery and sediment deposition in reservoirs (Brown et al., 2009; Foster, 2006; Verstraeten and Poesen, 2002).

The siltation of reservoirs is one of the most severe problems in the semi-arid environment of Burkina Faso (Cecchi et al., 2009). Due to the annually unevenly distributed and often variable rainfalls, many small dams have been constructed to store rainfall and runoff water in the rainy season for domestic use, irrigation, and stock watering in the dry season. As potential sinks for upstream sediments, dammed reservoirs are very prone to siltation. Accumulated soil particles may lead rapidly to changes in reservoir bed morphology, reduced water storage capacity and diminished water use potential. Especially shallow reservoirs in flat terrains are affected by comparably high losses in water volume due to both a relatively fast increase in the accumulation layer at the bottom of the reservoir and high evaporation losses from a comparably large water surface area. On the other hand, reservoir investigations in Africa have shown that small reservoirs offer a reliable tool to estimate soil deposition rates and long-term sediment yield from the surrounding catchment (Albergel et al., 2004; Gresillon and Reeb, 1981; Haregeweyn et al., 2008; Liebe, 2005; Tamene et al., 2006*)*.

In this study, siltation rates of three small reservoirs in southwestern Burkina Faso are analyzed firstly by bathymetric surveys in order to identify changes in reservoir bed morphology, and secondly by sediment cores in order to determine the magnitude of accumulated sediment. The specific objectives of the reservoir investigations are i) to calculate changes in water storage capacity by comparing initial and actual stage-storage and stage-area curves of the reservoirs, ii) to quantify the thickness of accumulated sediments along a longitudinal cross-section of the reservoir bed, and iii) to compare stratigraphic changes and down-core variations in sediment properties with estimates from the bathymetric survey.

Furthermore, the ^{137}Cs technique is used as an independent and supplementary method to validate the results of the bathymetric survey. In addition to its well-established application for soil erosion assessment at hillslope scale (Chapter 3), the ^{137}Cs technique provides a valuable means to calculate sediment yield in reservoirs (Foster and Walling, 1994; Foster et al., 2007; Simms et al., 2008). As about 30-40 % of the original radionuclide fallout still remains in the soil and has been found detectable in the environment of Burkina Faso (section 3.5.3 and 3.5.4), ^{137}Cs can be used as a tracer for transported and deposited sediments. As reservoirs present major sink areas, the concentration of ^{137}Cs in reservoir sediments is comparatively high. Depth profiles of ^{137}Cs cores can show significant changes in ^{137}Cs concentrations between the topsoil and the subsoil, indicating that the in-situ reservoir bed is reached. The ^{137}Cs approach is therefore often applied in combination with other sedimentological records to determine variations and abnormalities in deposition rates (Brown, 2009; Zhang et al., 2005). In Burkina Faso, undisturbed sediment samples from the bottom of the reservoir are used to analyze the magnitude of ^{137}Cs in reservoir beds and to detect its variations with depth. The specific objectives of the ^{137}Cs technique are i) to differentiate between accumulated allochthonous sediment and in-situ autochthonous soil in order to determine the sediment thickness of the reservoir, and ii) to compare these results with the stratigraphical records from the former analysis of down-core variations in soil properties.

Finally, the soil erosion and sediment delivery model WaTEM/SEDEM is used to simulate average annual soil loss and deposition rates at a catchment scale. The spatially distributed model allows analysis of on-site and off-site impacts of soil erosion by predicting sediment production and sediment delivery, spatially explicit for each raster cell of the catchment (Van Oost et al., 2000; Van Rompaey et al., 2001; Verstraeten et al., 2002). Although the model has been mainly applied for catchments in Europe (de Vente and Poesen, 2005; Keesstra et al., 2009; Verstraeten and Prosser, 2008; Ward et al., 2009), it has been adapted to various environments outside Europe by calibrating the transport capacity coefficient of the model (Feng et al., 2010; Li et al., 2007; Tiessen et al., 2009; Verstraeten et al., 2007). As the model also predicts sediment deposition in ponds/reservoirs, it can be validated against measured deposition values from the bathymetrical survey. Besides the prediction of total values for sediment production, export and deposition, a spatial map of erosion/deposition rates for each cell of the catchment is generated which allows identification of the most affected soil erosion hazard zones. These hazard zones present catchment areas where the tolerable soil loss rate is exceeded, which implies that the soil and its functions are no longer maintained, and that soil formation and soil loss are no longer balanced (Li et al., 2009; Roose, 1994; Verheijen et al., 2009). Such an erosion hazard map is considered a useful tool to select the most suitable locations for implementing soil and water conservation measures and to define a threshold value for policy interventions (Volk et al., 2010).

The specific objectives of the modeling approach are i) to predict total annual sediment production, delivery and deposition rates for the catchments, ii) to compare simulated against measured (specific) sediment yields from the bathymetric survey, and iii) to develop an erosion hazard map showing the most severely affected soil loss and soil accumulation zones (sediment source and sediment sink areas) and present a first soil loss tolerance threshold.

4.2 Site description

The studied catchment is located in the Ioba province of southwestern Burkina Faso and drains in the Mouhoun (also known as Black Volta River), which belongs to the watershed of the Upper Volta Basin in West Africa (Figure 4.1). Three sub-catchments were selected within this larger catchment. All are headwater catchments characterized by ephemeral flows draining in small reservoirs behind a constructed dam.

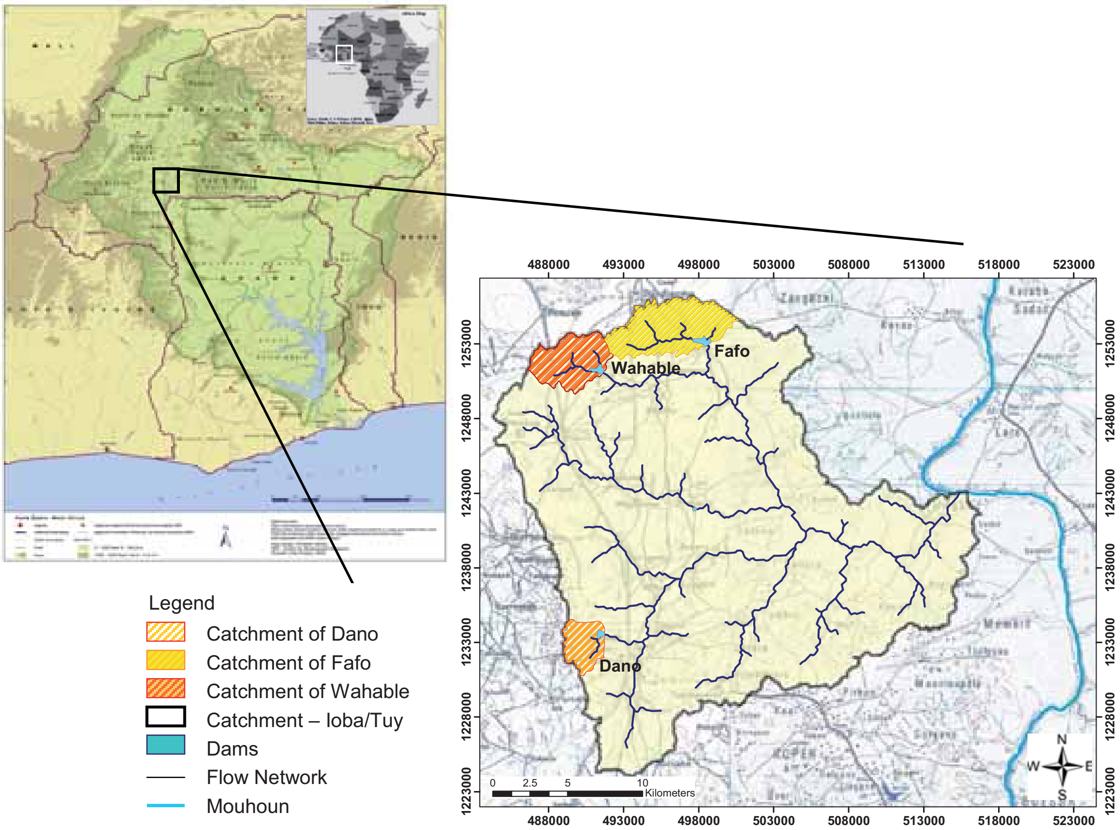


Figure 4.1 Location of study catchments in the Ioba province, southwestern Burkina Faso

The three sub-catchments are:

1) The sub-catchment of Dano, located at latitude 11°16’ N, longitude -3°08’ W and at 300 m asl in the administrative department of Dano (Ioba province). The dam (“Barrage du Moutori”) was built in 2001/2002, has a maximum pool level of 720 x 10^3 m^3 and a water surface area of 24 ha, and retains water and sediments from an upstream area of approx. 8 km^2.

2) The sub-catchment of Wahable, located at latitude 11°32'N, longitude -3°08 W and at 295 m asl in the administrative department of Oronkua (Ioba province). The dam ("Barrage de Wahable") was built in 1988, has a maximum pool level of 680 x 10^3 m^3 and a water surface area of 34 ha, and retains water and sediments from an upstream area of approx. 15 km^2.

3) The sub-catchment of Fafo, located at latitude 11°33'N, longitude -3°01 W and at 273 m asl in the administrative department of Koti (Tuy province). The dam "Barrage de Fafo" was built in 1987/88, has a maximum pool level of 870 x 10^3 m^3 and a surface area of 44 ha, and retains water and sediments from an upstream catchment area of approx. 24 km^2.

The larger catchment covers a total area of 580 km^2 and has a maximum north-south and west-east extent of 30 km. Due to the low spatial distance, all sub-catchments belong climatically to the Sudano-Sahelian zone, which is characterized by a mono-modal rainfall pattern from May to October with approximately 950 mm annual rainfall (Figure 3.1, section 3.2). Agro-ecologically, the area is part of the Sudanian domaine with arboraceous and scrubby savannah vegetation, abundant perennial grassland, but also sparse forest areas and riparian vegetation along ephemeral flows. Annual crops such as millet (*Pennisetum glaucum*), sorghum (*Sorghum bicolour*), maize (*Zea mays*) and cowpeas (*Vigna unguiculata*), and more recently also cash crops such as cotton (*Gossypium herbaceum*), are increasingly replacing the natural vegetation. Irrigation agriculture is also gaining importance, and rice (*Oryza sativa*) and various vegetables are cultivated near artificial and natural reservoirs.

The ancient Precambrian peneplain, a vast and monotonous plateau of crystalline and metamorphic rocks, determines the low average slope gradients of 2-3° and has an average altitude of 300 m asl. The plateau is slightly declining to the east and has an altitude of 240 m asl at the inflow to the Mouhoun. The natural boundaries of the catchment are some lateritic hills in the north and south and the Ioba Mountains in the west.

The predominating soils are lixisols (WRB, 1998), which have a low nutrient status, low organic matter content and a sandy loam to sandy clay loam structure. These highly weathered and leached tropical soils also cover the main area of the sub-catchments of Wahable and Fafo. More fertile cambisols (WRB, 1998) with a higher soil nutrient status and organic matter content and a clay to silty clay texture can be found in the western part (e.g., sub-catchment of Dano), where they cover small patches mainly on the hillslopes. Gleysols and fluvisols occur mainly in the lowland valleys, along water flows and in the flood plains of the Mouhoun. The spatial distribution of soil types is highly related to the topography and hence their position in the landscape. The interaction of pedo-geomophic processes and soil development along a hillslope (see section 3.5.1) can be also applied to a larger landscape unit such as a catchment.

The catchments belong administratively to the Ioba province – except the northern sub-catchment of Fafo, which is part of the Tuy province. The Ioba province has 200000 inhabitants, and more than 90 % of the population lives in rural settlements. Beside the largest town Dano with approximately 10000 inhabitants, most of the widely spread rural villages have between 1000 and 2000 inhabitants. The growth rate of 2.6 % and the high migration of other ethnics, especially Mossis, into a Dagara-dominated area, led to a population increase of 20 % from 1996 to 2006 (INSD, 2007). An average population density of 60 inhabitants per km^2 is relatively high for Burkina Faso, and is reflected in the importance of and increasing demand for land.

The construction of dams and the emergence of semi-natural reservoirs, often as a result of road construction, improved the access to water and facilitated irrigated agriculture in the dry season. Until the year 2001, more than 850 dams and semi-natural reservoirs were recorded by the national census for Burkina Faso. Most of the reservoirs were constructed in the 1980s and are very small. Approximately 80 % of the reservoirs have a water storage capacity below $1000 \times 10^3 m^3$ and 75 % of these have capacities below $400 \times 10^3 m^3$ (Figure 4.2). Cecchi et al. (2009) even estimated a total number of about 1650 reservoirs in 2009 in Burkina Faso.

Besides a large number of semi-natural reservoirs, about 18 dams were constructed in the Ioba region and, except for three of them, were all built in the 1980s or later, probably due to the difficult access by road to this remote region. The studied small reservoirs of Dano, Wahable and Fafo are representative for the region in terms of size, age and environmental characteristics (e.g., soil, vegetation, hydrological parameters) and at the same time show enough similarity and sufficient differences to allow comparative studies (Table 4.1).

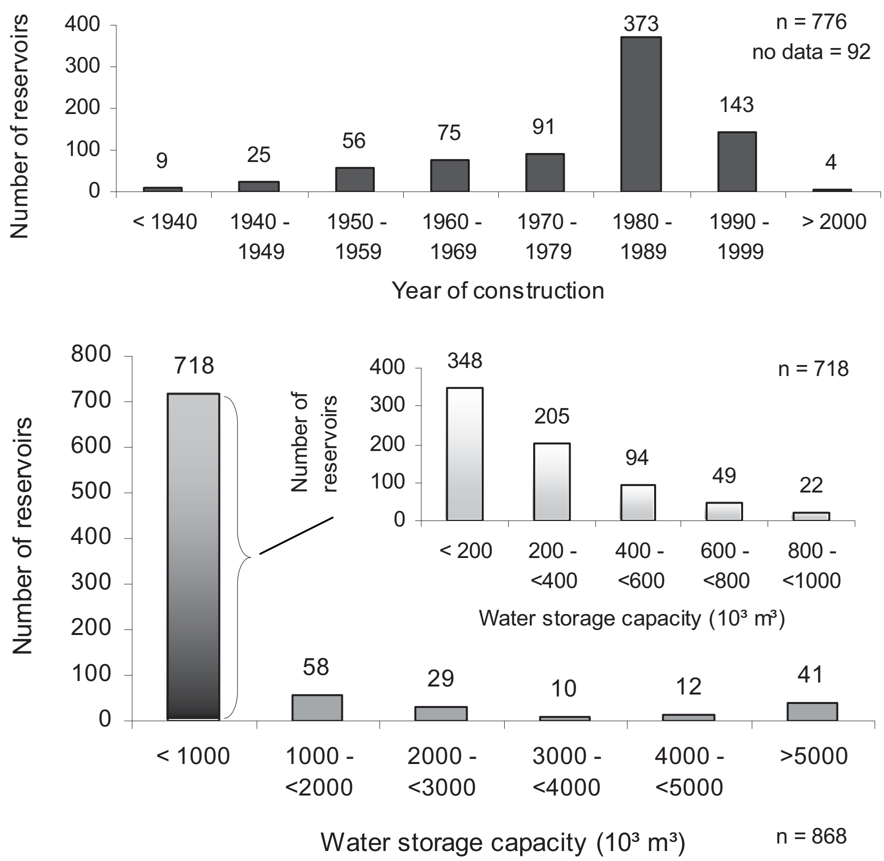


Figure 4.2 Year of construction and water storage capacity of reservoirs in Burkina Faso (Source: BDOT, IGB, 2001)

The dam of Dano, financed by the Dreyer foundation, is an example of a recently built dam presenting an early sedimentation stage of a small reservoir and a dam in an uncommonly accentuated environment with slope gradients of up to 25°. The small sub-catchment is surrounded by the Ioba Mountains, a Birrimian hillslope chain from the mid-Precambrian, with a maximal elevation of 502 m asl. The comparatively steep rise of the hillslopes of more than 200 m above average elevation provokes a high relief energy and hence high runoff rates. As the lateritic and clayey soils have low infiltration rates and a high surface sealing, approximately 10-20 % of the rainfall is expected to become runoff. Ephemeral flows and emerging gullies of a few meters width and two to three meters depths give evidence of the eroding impact of runoff due to the high relief energy, and imply a high sediment delivery rate. Two lateritic plateaus south and north of the valley enclose the small catchment like a bottleneck where the dam is located. The dam crest has a length of 277 m, a height of 9 m, and the spillway is 12 m wide.

Table 4.1 Characteristics of small reservoirs at the study sites

Specific data	**Dano**	**Wahable**	**Fafo**
Name	Barrage du Moutori	Barrage de Wahable	Barrage de Fafo
Department	Dano	Oronkua	Koti
Province	Ioba	Ioba	Tuy
Longitude	-3°08’	-3°08’	-3°01’
Latitude	11°16’	11°32’	11°33’
Altitude (m)	306	301	278
Year of initial topographical survey	April 1999	August 1984	June 1987
Year of construction	2001/02	1988	1987/88
Inauguration of dam	October 2002	August 1988	November 1988
Constructed by	SHW, ONBAH	ONBAH	ODE /FEME
Catchment area (km²)	7.9	14.8	23.6
Max. pool level (10 m³)	720	680	870
Surface area (ha) at max. pool level	24	34	44
Water depth (m) at max. pool level	7.5	4.8	4.8
Water capacity at normal pool level (10 m³)	360	480	480
Surface area (ha) at normal pool level	16	29	32
Water depth (m) at normal pool level	5.7	4.2	3.8
Dam height (m)	9	6.6	5.5
Dam crest length (m)	277	444	937
Spillway length (m)	12	4	50
Length of main tributary (km)	3.5	5	6
Actual irrigated land (ha)	20	7	15
Population number of village/ Potential water users of surrounding villages	10000	1400 / 100000	2060 / 12000
Main ethnics	Dagara	Dagara	Dagara, Bobo, Mossi, Pougli, Peulh

Sources: ODE, 1989; ONBAH, 1984; PNG-2, 2004; Wittenberg, 1998.
SHW: Sozietät für Hydrologie und Wasserwirtschaft; ONBAH: Office National Des Barrages et Des Aménagements Hydro-Agricoles; ODE:Office de Développement Evangélique; FEME: Fédération des Eglises et Missions Evangéliques

The catchment has a contributing area of less than 8 km² and a tributary stream of approximately 3.5 km (Wittenberg, 1998). Although the small size of the contributing area is the determining factor for the potential rainfall and runoff volume filling the reservoir, the water volume was, at least in rainy years, sufficient to ensure irrigation of approximately 20 ha in the dry season. However, the normal pool level at which the spillway is flooded has never been reached, and the reservoir falls dry in years with low annual rainfall (e.g., at the end of the dry season in March/April 2006).

In contrast to Dano, the sub-catchments of Wahable and Fafo can be seen as more representative for the typically semi-arid environment of the Burkina Faso plains with average slope gradients of 2-3° and steepest slopes gradients of 10° on small lateritic hilltops. Approximately 5 % of the annual rainfall is expected to reach the reservoir from runoff from the contributing area. Both dams were built in 1988 by national engineering agencies to improve the access to water for domestic and animal use and to overcome water shortage problems, which had led to a high livestock mortality (especially cattle) during a severe drought in 1983/84.

The small reservoir of Wahable has an all-year depth of at least 2-3 m and a small surface area bordered by a slightly rising lateritic shoreline, which hinders a flooding of large surrounding areas. The catchment has a medium drainage density of 0.97 km per km². The main tributary is approximately 5 km long and shows clear signs of regressive erosion and related erosive as well as accumulation processes. The number of potential water users is 11400 inhabitants when considering the village of Wahable as well as the five surrounding villages within a distance of 8 km that have access to the reservoir. Besides for domestic use, the water is mainly required for irrigated agriculture and livestock. About 2000 cattle and numerous sheep and goats from the surrounding region use the reservoir as watering place in the dry season (ONBAH, 1984). The annual consumption of water from the reservoir is estimated to be 100000 m³ for irrigation agriculture in an area of approximately 7 ha, 80000 m³ for domestic use, and 20000 m³ for livestock.

In contrast to Wahable, the reservoir of Fafo is characterized by a large surface area of 44 ha but a low and annually strongly fluctuating water depth resulting in a seasonal flooding of vast areas. The main tributary has a length of 6 km and shows only slight signs of regressive erosion. The extent of the reservoir area is also expressed by the length of the dam crest of 940 m and the width of the spillway of 50 m. The extensive water surface leads to serious water storage losses due to evaporation processes. These losses can reduce the total water volume by up to 60 %, which can lower the overall water depth by 2 m. Additionally, the storage volume is diminished by water consumption for domestic use, livestock and irrigation, and a minimum water capacity of 450000 m³ is needed to retain the reservoir function and to provide adequate water capacities in the dry season. Similar to Wahable, a large number of potential water users exist when the surrounding villages (12000 inhabitants), cattle (1100) and sheep (800) in the region are included (ODE, 1989). The estimated annual water consumption for domestic use is 84000 m³, for livestock 14000 m³, and for irrigated agriculture approximately 200000m³ when assuming mixed farming of rice and vegetables on an irrigated area of 15 ha.

4.3 Data and methods

4.3.1 Topographical survey

The Shuttle Radar Topography Mission (SRTM) provides digital elevation data with a resolution of 90 m x 90 m for West Africa. As the resolution of the SRTM-dataset was too low, a subset of the SRTM-elevation model was smoothed, resampled and interpolated by the spline routine in ArcView to a smaller cell size of 15 m x 15 m. In order to downscale the resolution of the SRTM, elevation data with a higher resolution were collected for the catchment area by a Differential Global Positioning System (DGPS, Ashtech Locus survey system) in 2006.

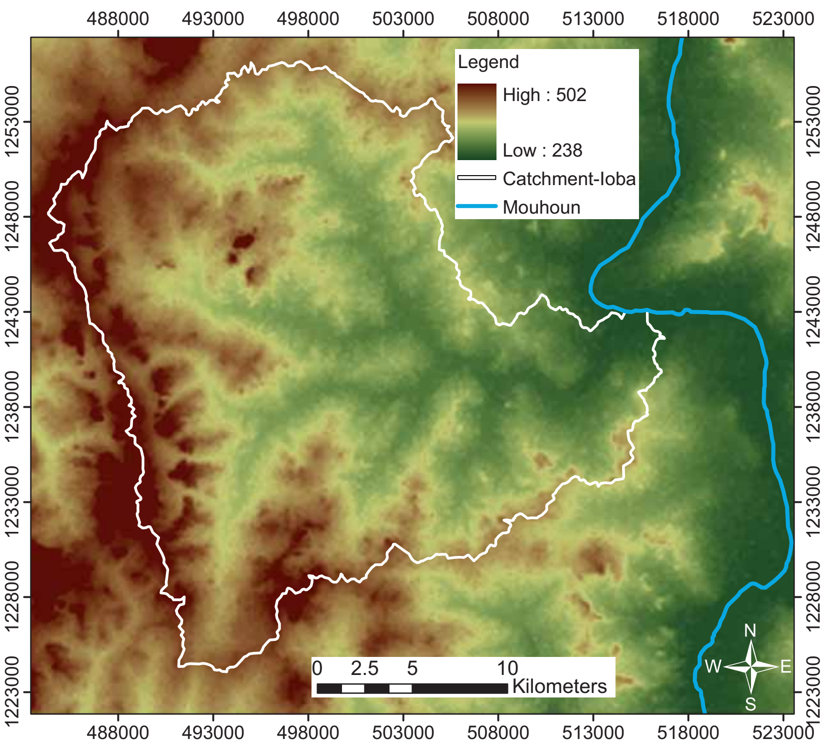


Figure 4.3 Digital elevation model for the catchment in southwestern Burkina Faso

A national control point provided by the Geography Institute of Burkina Faso (IGB) served as a reference point for the base station. A total of 40 measured DGPS points were processed in GNSS-Solution software. All geographic information was projected in UTM WGS 84, Zone 30 North (Figure 4.3). An accuracy assessment showed an RMSE of 1.02 m in elevation between DGPS measurements and downscaled elevation data.

4.3.2 Meteorological and hydrological data

Meteorological data for the total catchment area are based on available daily rainfall data from the Regional Agricultural and Hydrological Service of the Ioba region (DPAHRH, Burkina Faso), which provided reliable rain-gauged data for Dano. More specific, continuous daily data on temperature, air pressure and rainfall intensities (10-min. intervals) were available for the year 2006 from a climate station in Dano installed by the Glowa-Volta Project (ZEF, University of Bonn).

Additionally, rain gauges (type Hobo, Micro Station) were located at each sub-catchment from May 2005 to February 2007 to account for regional and small-scale variations in rainfall. For the same time period, water level and temperature data loggers (Madgetech, Level 1000) were installed permanently in each reservoir to record changes in water levels and to evaluate the water retention pattern of the reservoirs. As the water head is expressed in pressure units (1 PSI correspond to 68 948 hPa), the values were corrected for atmospheric pressure.

4.3.3 Bathymetric survey

The bathymetric approach is based on a comparison of changes in reservoir morphology at two different times, i.e., at the time of the initial topographical survey before the dam was constructed (Dano: 1999; Wahable: 1984; Fafo: 1987), and at the time of the bathymetric survey (Dano: 2006; Wahable: 2005; Fafo: 2006).

The initial topographical map of the reservoir (resolution 1:1000) and the corresponding geo-technical report provided the required information about area-volume curve, potential water storage capacity, dam height and length, spillway length and overflow characteristics as well as construction details, such as degree of surface soil disturbed or sediments removed. As the topographical reservoir map was not georeferenced, a total of 840, 261 and 574 elevation points were digitized in ArcGIS for the reservoirs of Dano, Wahable and Fafo, respectively, and assigned to the reference system UTM WGS 84, Zone 30 North. For each map, a digital elevation model (DEM) was generated in Surfer (Golden Software, version 8.0) using point kriging.

The actual morphometry of the three reservoirs was measured either directly by a topographical survey using a tachymeter (Leica, type Wild NK-1 non-automatic tilting level) or indirectly by a bathymetric survey by water-depth measurements using a depth sounder with a mapping GPS unit sonar (Lowrance, LMS-480M).

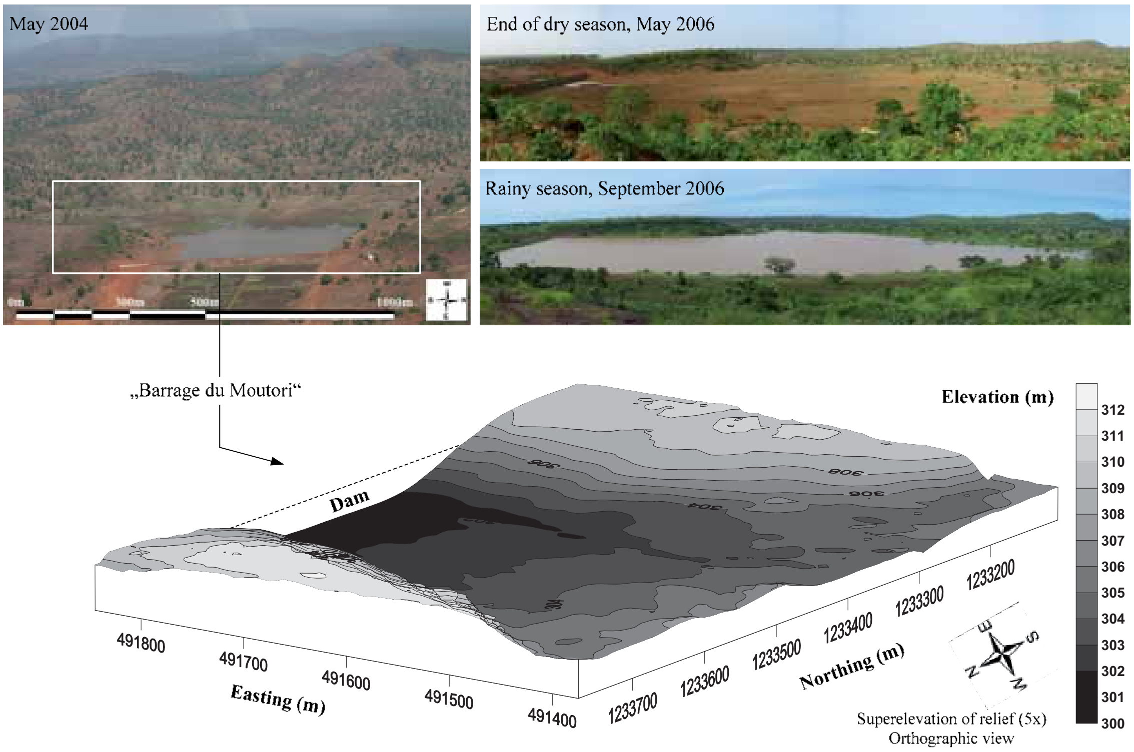


Figure 4.4 Digital elevation model of the reservoir of Dano

The direct approach was used for the research site in Dano when the reservoir had fallen completely dry in May 2006. A total of 2064 topographical points were collected in a spatial grid of 10 m by 10 m and processed by point kriging in Surfer software to generate a DEM (Figure 4.4). The overall accuracy of the DEM was determined by an independent subset of 75 measured elevation points. The differences in elevation between gridded and measured values show a root mean square error (RMSE) of 0.08 m.

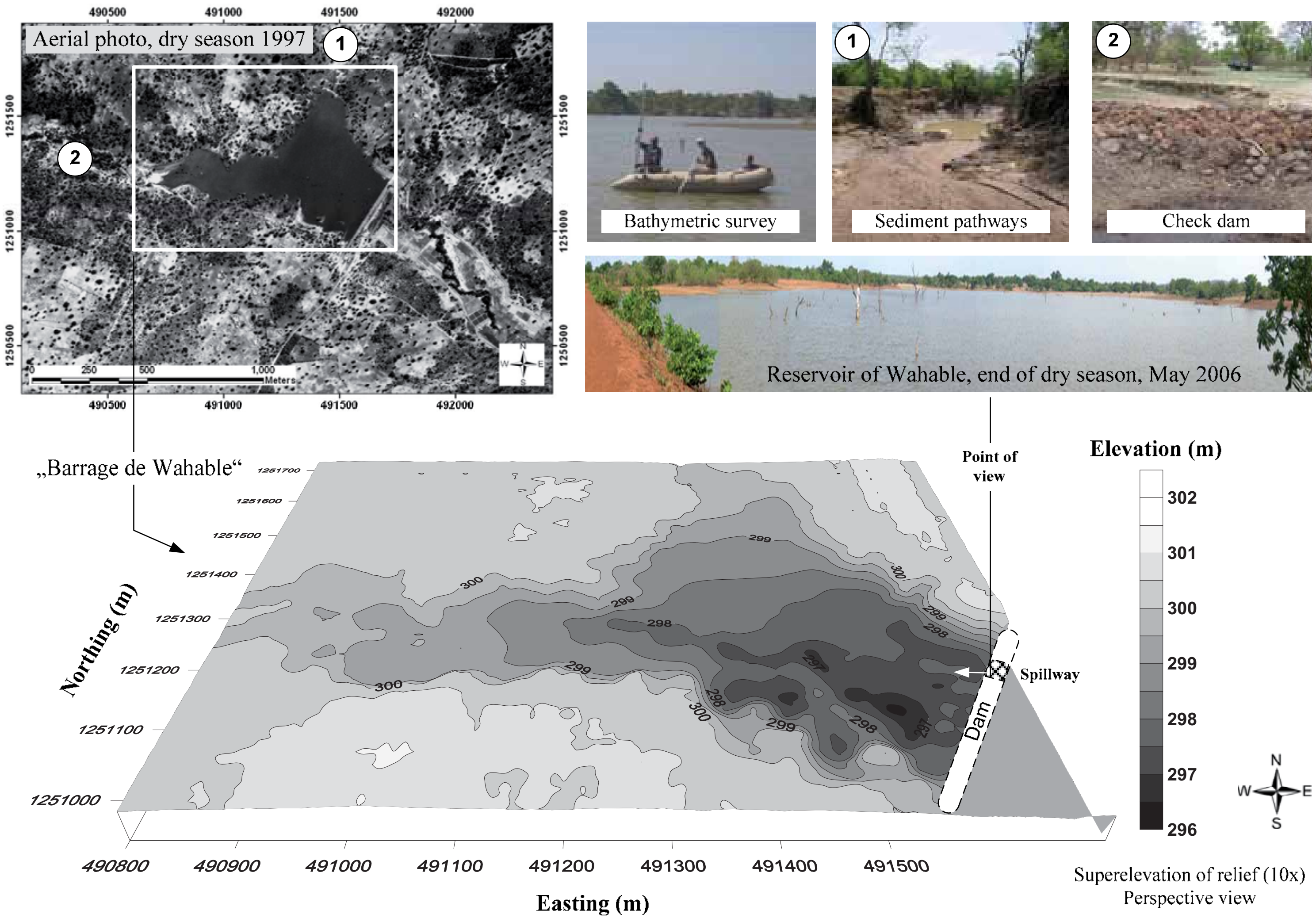


Figure 4.5 Digital elevation model of the reservoir of Wahable

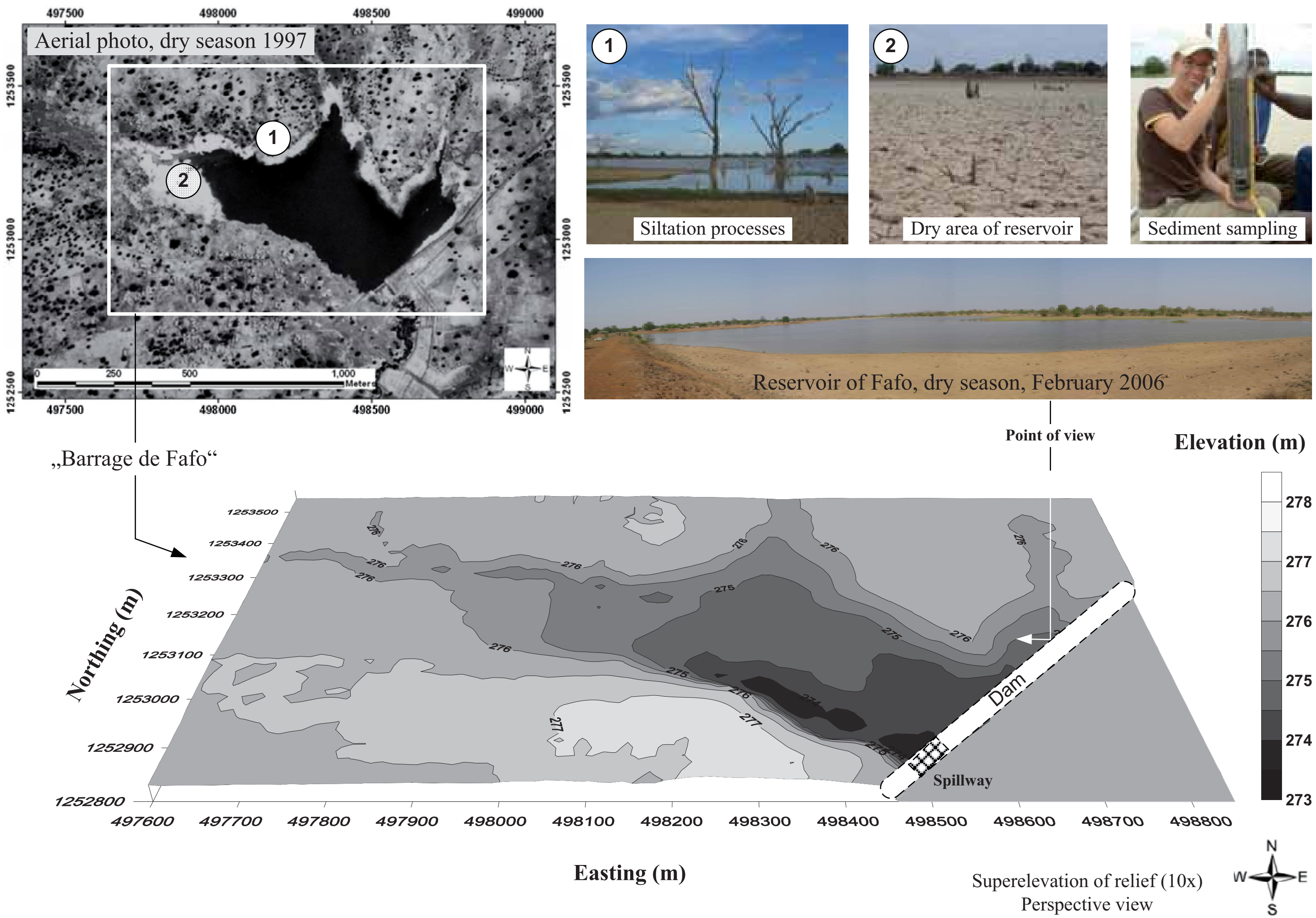


Figure 4.6 Digital elevation model of the reservoir of Fafo

The indirect, bathymetric survey was applied for the reservoirs of Wahable (December 2005) and Fafo (June 2006), which were permanently filled with water. For the survey, the sonar depth sounder was attached to an inflatable boat, and a total of 1104 and 664 points were collected for Wahable and Fafo, respectively, in a spatial sampling grid of approximately 25 m by 25 m. The water-depth measurements were crosschecked by a simple stadia rod thus ensuring a high accuracy especially for shallow depths below 0.3 m and the shorelines. All water-depth points and their corresponding GPS coordinates were processed by GIS-software (MapCreate, ArcView, ArcGIS). The final bathymetric map was generated by point kriging in Surfer software (Figure 4.5 and 4.6).

Based on an independent subset of 50 measured depth points at both sites, the overall accuracy in height between gridded and measured data was determined by the standard error of prediction. The RMSE is 0.14 m in elevation for Wahable and 0.09 m for Fafo.

For comparative analysis between the initial and actual morphology of the reservoirs, an independent reference point was selected outside the reservoir that had not experienced sedimentation or erosion (e.g., a point on a bold, lateritic plateau near the shoreline). Based on this fixed reference point, the relative elevation values of the initial topographical maps were adjusted to the absolute height values of the recently measured values. For each contour level, the initial and actual water volume and the corresponding surface area were calculated by three numerical integration algorithms in Surfer, namely the trapezoidal rule, the Simpson's rule and the Simpson's 3/8 rule (Press et al., 1988). The results show a close agreement between each method suggesting a high accuracy in volume calculations and a sufficiently high grid density. By knowing the differences in initial and actual storage volume, changes in water storage volume could be quantified.

4.3.4 Lake core sampling by Beeker sampler

The coring of sediments from the bottom of the reservoir was found to be a feasible approach to quantify the thickness of the accumulated sediment layer, and offered at the same time a valuable method to validate results obtained from other methods (e.g., bathymetric survey) with single point data serving as reference values.

If the reservoir or at least a large part of it dried up seasonally (e.g., in Dano) soil pits were dug, and the thickness of accumulated sediment was determined directly by measuring the increment in the upper soil layer.

If the water level was too high all the year round to take soil samples from soil pits (e.g., in Wahable and Fafo), sediment cores were removed from the bottom of the reservoir by underwater equipment from an inflatable boat. The sediment core sampler, type Beeker sampler (Eijkelkamp), was used to obtain undisturbed soil samples of up to 1 m length from below a water depth of 2-3 m. The Beeker sampler consists of a rodding frame in which a transparent tube of 1 m is inserted. A piston on a rod is twisted into the transparent tube and fixed at the cutting head of the sampler. The inflatable membrane at the inside of the cutting head is connected by an extension hose with a pressure pump. To assure an accurate vertical penetration of the core sampler and a successful withdrawal of the sediment, the boat should be stabilized with several iron rods pushed into the ground. Then the sediment sampler is lowered until the solid bottom sediment (nautical depth) is reached. While pushing and/or hammering the cutting head into the sediment, the cord of the piston is fixed to the stable boat so that the piston remains at the same height while the sampler and its tube are pushed downwards. Once the required depth is reached, the inner membrane of the sediment head is inflated by 2-3 bars pressure closing the core opening to prevent the sediment from dropping out of the sampler. The sampler is pulled up slowly into the boat, and the sediment thickness in the transparent tube is recorded. After deflating the inner membrane, the sediment can be removed in intervals of a few centimeters from the transparent tube.

For the analysis of physical and chemical soil properties, two composite samples were taken for each reservoir by a Beeker sampler at a distance of 2-3 m from each other to a soil depth of 80-100 cm. Sediment samples of 5 cm depth increments were analyzed in a soil laboratory for organic matter, bulk density, soil texture, pH, carbon (C total), nitrogen (N total), phosphorus ($P_{available}$), potassium ($K_{available}$), cation exchange capacity (CEC_{pot}) and exchangeable cations (for methods see section 3.3.3).

4.3.5 ^{137}Cs measurements in reservoirs

The concentration of ^{137}Cs in reservoir sediments was either measured directly by an undisturbed soil sampler (split tube sampler, Eijkelkamp) when the reservoir was dry (in Dano) or indirectly by a sediment core sampler (Beeker sampler, Eijkelkamp) when the reservoir was full of water (in Wahable and Fafo). Whereas the undisturbed sediment samples had a diameter of 4.8 cm², a surface area of 18.1 cm² and a volume of 90.5 cm^3 (considering a soil depth of 5 cm), the sediment core sampler had a diameter of 5.6 cm, a surface area of 24.6 cm² and a volume of 123 cm³ (considering a soil depth of 5 cm).

First, five composite samples at a distance of about 80 m were collected along the longitudinal axes of the reservoirs to determine the spatial distribution of ^{137}Cs and to estimate typical accumulation rates. At each of the five sampling points, five bulk composites were taken about 10 m apart from each other. A total sampling depth of 30 cm for composite samples was chosen to facilitate a better comparison with reference and hillslope samples and to consider the deposited sediment in the reservoirs of Wahable and Fafo.

Second, a depth soil core was taken at the point of the highest ^{137}Cs concentration, which at all locations was the sampling point closest to the dam. The sediment core had a total depth of 60 cm, 80 cm and 100 cm for the reservoirs of Dano, Wahable and Fafo, respectively, and was divided into 5 cm depth increments to account for the variability of ^{137}Cs with increasing depth. In order to both reduce the spatial variability of individual sampling points and to gain a sufficient amount of soil for laboratory analysis, two composite samples approximately 3 m apart from each other were combined.

The areal activity (Bq m^{-2}) was calculated based on the equation of Sutherland (1994), which takes the dry bulk density of the fine fraction into account. The depth distribution of ^{137}Cs concentration (Bq kg^{-1}) and the ^{137}Cs areal activity in the reservoirs were compared with the depth distribution of the local reference sample at a stable site, here the Natural Reserve of Bontioli. The magnitude of soil deposition was calculated by the ^{137}Cs conversion models (proportional model and mass-balance models 1 and 2) for each of the sampling points. Further details about the selection of the reference samples, the sampling procedure, the laboratory and isotope analysis as well as specifications of the conversion models are given in sections 3.3.4 and 3.3.5.

4.3.6 Calculation of sediment yield

The measured reservoir sedimentation rates from the bathymetric survey and the lake core sampling approach were used to calculate sediment yield for the three sub-catchments. The following equation was proposed by Verstraeten and Poesen (2002) to assess sediment yield and area-specific sediment yield from the corresponding catchments:

$$SY = 100\frac{SV \times dBD}{TE} \tag{4.1}$$

where SY = sediment yield (t yr^{-1}), SV = measured sediment deposition rate in volumetric units (m^3 yr^{-1}), dBD = dry bulk density of the sediment deposit (t m^{-3}), and TE = sediment trap efficiency of the reservoir (%).

The area-specific sediment yield (SSY) is calculated by dividing SY by the catchment area:

$$SSY = 100\frac{SV \times dBD}{TE \times A} \tag{4.2}$$

where SSY = area specific sediment yield (t $ha^{-1}yr^{-1}$), and A = catchment area (ha).

All required parameters for the equation were gained directly from reservoir investigations. Representative depth samples for dry bulk density (dBD) were taken for each reservoir from highly saturated sediment deposits by a Beeker sampler or directly from soil pits. The depth samples were subdivided into 5 cm depth increments, weighed and sieved and analyzed for dry bulk density and soil texture distribution (section 4.3.4). Considering the total accumulation depth for each reservoir, the weighted average of the individual depth increments was determined.

In order to convert the volume of the sediment samples into mass, the average dry bulk density of each sample, also known as specific or unit weight of reservoir deposits, was calculated by taking the operational condition of the reservoirs into account. The initial specific weight was based on the empirical equation developed by Lara and Pemberton (1963) (in Morris and Fan, 1997, p. 10.30):

$$W = W_C \times P_C + W_M \times P_M + W_S \times P_S \tag{4.3}$$

where W = specific weight of the deposit, P_C, P_M, P_S = percentages of clay, silt and sand in the sediment, and W_C, W_M, W_S = initial weight coefficients for clay, silt and sand in the sediment.

The initial weight coefficients for soil texture fractions vary with the operational conditions of the reservoirs (Table 4.2). Dano was periodically drawn down, whereas Wahable and Fafo were continuously submerged.

Table 4.2 Coefficients for initial sample weight in relation to grain size and operation conditions of the reservoir (Lara and Pemberton, 1963; in Morris and Fan, 1997, Table 10.3)

Operation condition	**Initial weight (kg m^{-3})**		
	W_C (Clay)	W_M (Silt)	W_S (Sand)
Continuously submerged	416	1120	1550
Periodic drawdown	561	1140	1550
Normally empty reservoir	641	1150	1550
Riverbed sediment	961	1170	1550

The trap efficiency (TE), defined as the proportion of incoming sediment that is deposited or trapped in the reservoir, represents one of the most important characteristics for sediment yield estimations (Verstraeten and Poesen, 2000). The empirical equation of Brown (1943) was used to calculate the average mid-term trap efficiency by considering the ratio of reservoir storage capacity to catchment area:

$$TE = 100 \times \left[1 - \frac{1}{1 + 0.0021 D \frac{C}{A}} \right] \tag{4.4}$$

where TE = trap efficiency (%), D = constant value ranging from 0.046 to 1 with a mean value of 0.1, C = reservoir storage capacity in m³, and A = catchment area in km².

The approach of Brown (1943) was chosen, as it is a simple but still commonly used and appropriate approach to estimate trap efficiency in small, semi-arid catchments, especially when long-term inflow and discharge data are not available. A constant value of 0.1 was chosen as the best fit for the small, semi-arid reservoirs in Burkina Faso (Table 4.3).

Table 4.3 Measured sediment yield (SSY and SY) and associated parameters

Specific parameter	Reservoir of Dano	Reservoir of Wahable	Reservoir of Fafo
Catchment area (A) (ha)	790	1480	2360
Reservoir storage capacity (C) (m^3)	360	480	480
Sediment deposition rate (SV) (m^3 yr^{-1})	2439	2000	1106
Dry-bulk density of sediments (dBD) (t m^{-3})	1.29	0.53	0.51
Trap efficiency (TE) (%)	91	87	81
Sediment yield (SY) (t yr^{-1})	3482	1212	696
Specific sediment yield (SSY) (t ha^{-1} yr^{-1})	4.4	0.8	0.3

4.3.7 Preprocessing of Aster images

Remote sensing techniques were used to provide a land-cover map to derive vegetation cover characteristics as input parameters for soil erosion modeling. Two satellite images from the Advanced Space-borne Thermal Emission and Reflection Radiomometer (ASTER) were available that covered the research area of the Ioba District along one flight path. The cloud-free images were taken at the end of the rainy season on 10 October 2004 just before harvesting time.

The two ASTER images have a swath width of 60 km x 60 km and a spectral, spatial and radiometric resolution of 15 m in the visible and near-infrared bands (VNIR, band 1-3), 30 m in the shortwave infrared (SWIR, band 4-9) and 90 m in thermal infrared bands (TIR, band 10-14) (Abrams and Hook, 2002).

For radiometric correction, the algorithm of ATCOR2 (Atmospheric and Topographic Correction, Version 2.0) implemented in ERDAS Imagine 8.6 was applied to reduce the influence of atmospheric and radiometric disturbance (Richter, 2002). The visible, near-infrared and shortwave infrared bands were used; the latter were resampled to 15 m resolution. From the thermal bands, only band 13 was integrated and adjusted to the other bands by rescaling it to the unsigned 8 bit format.

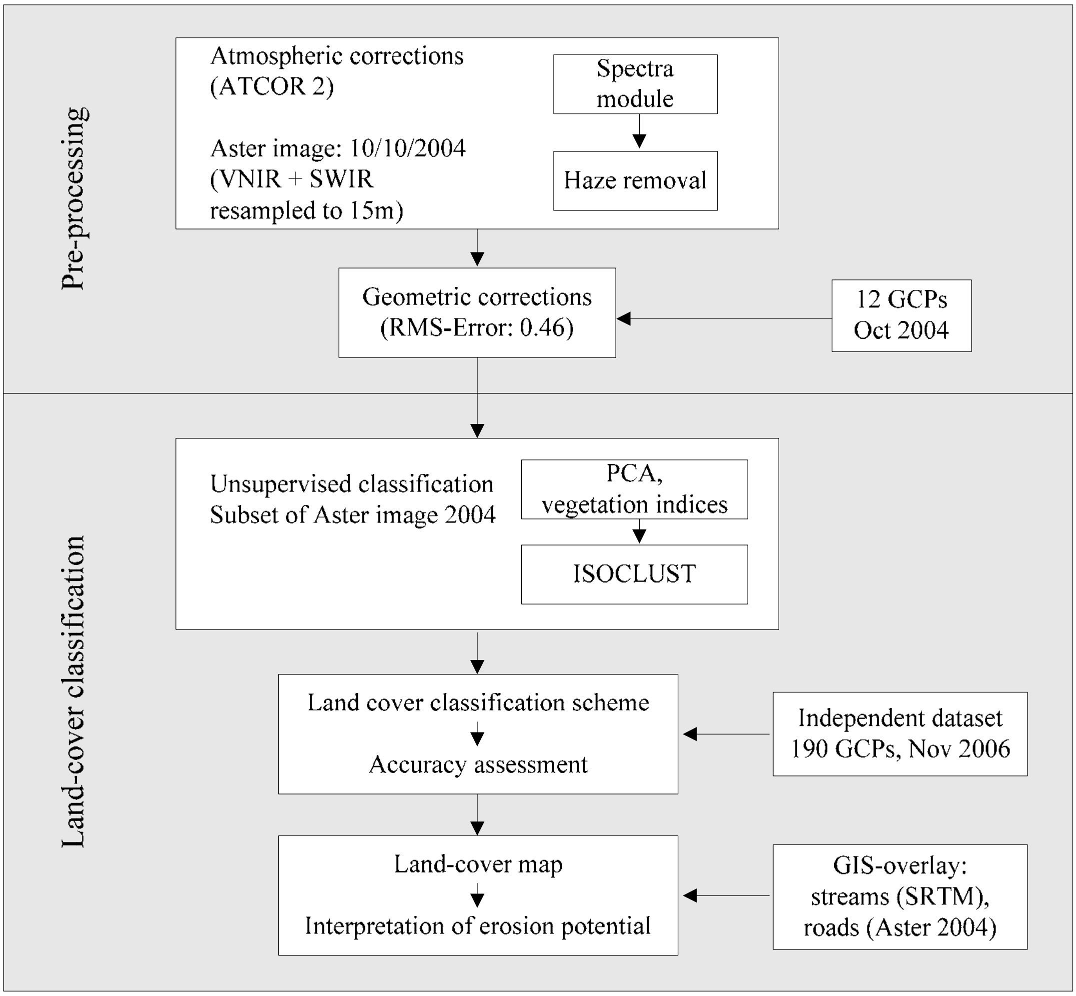


Figure 4.7 Pre-processing of satellite images and land-cover classification

This thermal band was required to facilitate and enhance the selection of spectral references curves within ATCOR, but was not used later for the land-cover classification.

The images were stacked, mosaiked and corrected for differences in sun position (solar zenith and solar azimuth). The radiance at sensor was calculated using gain state, offset values and conservation coefficients from an Aster calibration file. Atmospheric parameters (aerosol type and humidity) and scene visibility were adjusted within the spectra module of ATCOR2 by comparing reference reflectance curves against measured reflectance curves. The haze removal option was applied when reflectance values for different environmental features (e.g., water bodies, dry and sparse vegetation, concrete surfaces) matched within 10 % uncertainty level with reference values. When an agreement of 90 % spectral reflectance was reached, the specific coefficients were selected. All image processing and land-cover classification steps are shown in Figure 4.7.

Geometric correction of the Aster scene to UTM WGS 84, Zone 30 North reference system was performed using 12 ground control points (GCPs) from a field survey conducted in 2004 (05.-13.10.2004). Ground control points were collected by a handheld GPS (Garmin, eTrex) with a precision of 5-10 m at the same time period the Aster image was taken (10.10.2004). The RMS-Error (RMS 0.46; RMSx 0.34; RMSy 0.31) indicates subpixel (<15 m) precision of the geocorrection process. For resampling, the nearest neighbor method was applied.

4.3.8 Land-cover classification

An unsupervised classification was performed for a subset of the Aster image 2004 with IDRISI 32. The ISOCLUST-module in IDRISI functions as an iterative self-organizing unsupervised classifier and is based on the full maximum likelihood procedure to assign clusters (Eastman, 2001). For cluster analysis with ISOCLUST, different raw bands (here band 1-6), vegetation indices (here $TSAVI_1$), and a composite image were used.

For selecting the most appropriate vegetation indices, all distance-based vegetation indices were compared with respect to their efficiency in reducing the effect of sparse vegetation and interference between green vegetation and bare soil. A soil mask image was created using the Normalized Difference Vegetation Index (NDVI) for Aster Images: ASTER NDVI = (Band3n-Band2)/(Band3n+Band2). Finally, the vegetation index $TSAVI_1$ (Transformed Soil-Adjusted Vegetation Index) was selected as most appropriate to reflect the vegetation in the semi-arid research region (Baret and Guyot, 1991; Cyr et al., 1995; Flügel et al., 2003). The $TSAVI_1$ is expressed as:

$$TSAVI_1 = \frac{a((NIR - a)(RED - b))}{RED + a \times NIR - a \times b} \tag{4.5}$$

where NIR = reflectance in the near infrared band, RED = reflectance in the visible red band, a = slope of the soil line, and b= intercept of the soil line (Eastman, 2001, p. 96)

For the composite image, a principle component analysis (PCA) was performed to select the most appropriate raw bands. The PCA shows that band 3 as well as the short wave infrared bands contains valuable information correlating with the first and second component, which explains approximately 69 % and 18 %, respectively, of the variance. Therefore, bands 3, 5 and 6 were selected for the composite image. The composite image functions within ISOCLUST to derive clusters for the cluster seeding process. A total of 31 spectral classes could be differentiated from an automatically generated histogram. These classes were aggregated and re-classed in ArcView 3.2 following a land-cover classification scheme modified from project BDOT (Base de Donnée d´Occupation des Terres) for Burkina Faso (IGN-FI, 2005).

For accuracy assessment, all automatic classified pixels were compared with 190 known reference points from an independent dataset (Cord, 2007). These reference points were collected in November 2006 in the same season the earlier image of 2004 was taken. Differences due to classification schemes (FAO-LCCS by Cord, 2007 versus BDOT used in this study) could be standardized by redefining and adjusting the FAO dataset based on its detailed information about category, type and percentage of each land-cover class. The error matrix (Table 4.4) shows a relatively low overall accuracy of 0.64, which might be caused by the high heterogeneity of the vegetation cover in the study area.

Table 4.4 Error matrix

Unsupervised classification - Aster 2004 -	**Known reference data [a]** 101	103	104	105	106	108	Total
101 - Annual crops	30	3	3	5	1	9	51
103 - Open woodland/ river riparian vegetation	1	25	0	2	0	1	29
104 – Grassland	8	0	15	0	0	8	31
105 – Shrubland	6	4	2	20	3	3	38
106 - Settlement - artificial surface	1	0	0	0	10	0	11
108 - Rural settlement - bare soil	4	0	2	0	4	20	30
Total	50	32	22	27	18	41	190
Producer accuracy	0,60	0,78	0,68	0,74	0,56	0,49	1,00
User accuracy	0,59	0,86	0,48	0,53	0,91	0,67	1,00
KHAT (Kappa analysis)	0,55						
Overall accuracy	0,63						

[a] Independent dataset provided by Cord, 2007

The Khat value of 0.55 indicates that the observed classification is 55 % better than that resulting from a random assignment of pixels. Producer accuracy ranged from 0.49 (rural settlement with bare soil) to 0.78 (open woodland/river riparian vegetation); user accuracy ranged from 0.48 (grassland) to 0.91 (settlement with artificial surface).

A more differentiated view shows a relatively high producer accuracy for classes that are distinct from others, either due to a very dense vegetation cover (e.g., open woodland/ river riparian vegetation) or no vegetation cover at all (e.g., settlement with artificial surfaces). No confusion exists between these classes, which is especially important for the assignment of cover values for the subsequent erosion modeling procedure. In this context, the relatively low accuracies of the other cover classes should be interpreted by determining the classes they show confusion with. For example, the low user accuracy for grassland can be explained by the interference of this cover with annual crops and rural settlement/bare soil, which reciprocally explains the low producer and user accuracy for the other classes. This confusion is mainly caused by the heterogeneous, small-scale landscape pattern, where parcels of a few meters alternate with bare soil and rural settlement.

Environmental features, such as stream network and water surfaces (e.g., dams) as well as the road network, were digitized from a drainage map and Aster images, respectively, and then integrated as GRID-files of 1-2 pixels (15-30 m width) into the land-cover map.

4.4 Modeling approach

4.4.1 Description of WaTEM/SEDEM

The spatially distributed soil erosion and sediment delivery model WaTEM/SEDEM (Version 2004) was used to simulate annual soil loss and sediment fluxes into rivers and reservoirs on a catchment scale. The model is a combined version of two empirically based soil erosion models, namely WaTEM (Van Oost et al., 2000) and SEDEM (Van Rompaey et al., 2001), both developed by the Physical and Regional Geography Research Group K.U. Leuven, Belgium. The spatially explicit character and the raster-based structure allow division of the landscape into small spatial units, here raster cells. For each of the cells, sediment transport and deposition is calculated based on two components: Firstly, an adapted version of the empirical Revised Universal Soil Loss

Equation (RUSLE, Renard et al., 1997) to predict average annual soil loss rates, and secondly, a transport capacity equation to simulate average annual sediment delivery (Verstraeten et al., 2002).

The average annual soil loss rate was calculated by the adapted RUSLE using the 2D routing algorithm for complex 2D landscapes, as proposed by Desmet and Govers (1996):

$$A = R \times K \times LS_{2D} \times C \times P \tag{4.6}$$

where A = average annual soil loss per unit area (kg m^{-2} yr^{-1}), R = rainfall erosivity factor (MJ mm m^{-2} h^{-1} yr^{-1}), K = soil erodibility factor (kg h MJ^{-1} mm^{-1}), LS_{2D} = two-dimensional slope length factor, C = crop cover/management factor, and P = erosion practice factor.

The flux of sediment into the river network is defined by the sediment-routing algorithms and follows topographically derived flow paths. Sediment routing, in return, depends on the transport capacity influenced by rainfall erosivity, soil erodibility and topography. The transport capacity of the overland flow determines the average annual sediment delivery expressed by the following equation for each landscape cell (Van Rompaey et al., 2001):

$$TC = ktc \times R \times K(LS_{2D} - 4.1s^{0.8}) \tag{4.7}$$

where TC = transport capacity (kg m^{-1} yr^{-1}), ktc = transport capacity coefficient (m), and s = slope gradient (m x m^{-1})

If the local transport capacity is higher than the sediment flux, sediment is transported further downslope; if the local transport capacity is lower, sediment is deposited.

4.4.2 Input parameterization

All input files were generated as raster layers in IDRISI 32 with a resolution of 20 m x 20 m. The raster layers/maps include information about topography (LS-factor), rainfall erosivity (R-factor), soil erodibility (K-factor), crop cover and land-use/management (C-factors) for each of the three sub-catchments Dano, Wahable and Fafo. Erosion control practices (P-factor) were not taken into account, as soil and water conservation measures were not used catchment-wide and, moreover, often similar or smaller in size as the resolution of the raster cells, so that they could not be represented in the modeling approach.

The spatial coverage of each raster layer set was dependent on the catchment size of the specific sub-catchment and comprised an area of approximately 8 km², 15 km² and 24 km² for Dano, Wahable and Fafo, respectively. For model calibration, the transport capacity coefficient (ktc) was used.

Topographical map, river/runoff and pond maps

The topographical map was generated by a digital elevation model of the catchment area (section 4.3.1), from which the two-dimensional LS-factor could be automatically and internally calculated by the WaTEM/SEDEM model. In order to enhance the DEM, all pits were removed and a digital filtering option in IDRISI was applied (7x7 filter size) to allow smoother simulations. Furthermore, a river/runoff map was generated by the surface runoff accumulation function in IDRISI. Also, the location of sediment sinks and retention pools (here dams) was taken into account by a pond map, which was digitized from remote sensing images of the Ioba district. The integration of a pond map in simulations is important for comparing simulated versus measured sediment deposition behind dams (namely pond deposition) and calculating sediment yield.

Rainfall erosivity factor

The erosivity of rainfall (R-factor) is mathematically defined as a function of the kinetic storm energy and the maximum 30 min rainfall intensity. The R-factor presents the average annual erosivity for a location and should be based on rainfall data of at least 10 years (Wischmeier and Smith, 1978; Renard et al., 1994). Due to a lack of measured long-term records of rainfall intensities, the R-factor was estimated by the index suggested by Roose (1977, 1997, 2004) for precipitation in West Africa. He used annual rain gage records for more than 10 years and more than 20 meteorological stations in Côte d’Ivoire, Burkina Faso, Senegal, Niger, Chad and Cameroon to establish the following relationship between the average annual erosivity (R) and the average annual magnitude of rainfall (P):

$$R = [(0.5 \pm 0.05)P] \tag{4.8}$$

By applying this equation to long-term annual precipitation records over 20 to 50 years, Roose furthermore developed an isoerodent map for West Africa (Figure 2.1, Chapter 2).

When locating the sub-catchments of Dano, Wahable and Fafo on this isoerodent map, a R-factor of approximately 500 can be roughly estimated for southwestern Burkina Faso, which corresponds well with the more precise R-value of 478 calculated from Roose's equation based on an average annual rainfall of 956 mm for the Ioba-District. The latter R-value was used in the WaTEM/SEDEM model. As the model allows only a single value as input parameter for rainfall erosivity, this R-factor value is assumed to be representative for the three catchments, which each only cover a few square kilometres. Due to the metric units required by the model (namely MJ mm m^{-2} h^{-1} yr^{-1}), the original R-factor of the RUSLE was divided by 10000, and hence an input value of 0.0478 was entered into the model.

Soil erodibility map

Soil erodibility (K-factor) is a measure of the susceptibility or resistance of a soil to detachment and transport and depends mainly on soil texture and soil organic matter content.

The WaTEM/SEDEM model requires input data on soil erodibility either in the form of a single, representative K-value for the entire catchment or in the form of an erodibility map, which takes the different susceptibilities of soils into account. As the spatial variability of soils in the region is high, and its influence on erosion and runoff characteristics significant, an erodibility map was generated. This map was based on both measured soil data for the dominant soil types of the three sub-catchments and on available soil erodibility values for Burkina Faso. For measured soil data, the erodibility value was calculated by the equation proposed by Renard et al. (1997) taking soil texture, soil organic matter content, soil structure and permeability into account:

$$K = \frac{\left[2.1\times10^{-4}\left(12-OM\right)M^{1.14} + 3.25(s-2) + 2.5(p-3)\right]}{0.759} \tag{4.9}$$

where K = soil erodibility factor (kg h MJ^{-1} mm^{-1}), OM = organic matter content (%), M = texture product: %silt*(%silt+%sand), s = structure class, and p = permeability class.

All required input parameters were obtained from soil samples collected from representative hillslopes of the sub-catchments (Chapter 2; section 3.3.3). The K-factor was determined for each sample and then averaged for each soil type. In order to extrapolate these point/hillslope data to the entire Ioba catchment, the digital soil map of the national soil bureau in Burkina Faso (BUNASOL, 2000) was used (Figure 4.8). The dominant soil types served as a lead for the K-factor distribution. When for a specific soil type (here plinthosols) no measured data from field surveys were available, K-factor values were derived from detailed erodibility studies in West Africa (Roose, 1997, 2004; section 2.1.2) and assigned to those soil types.

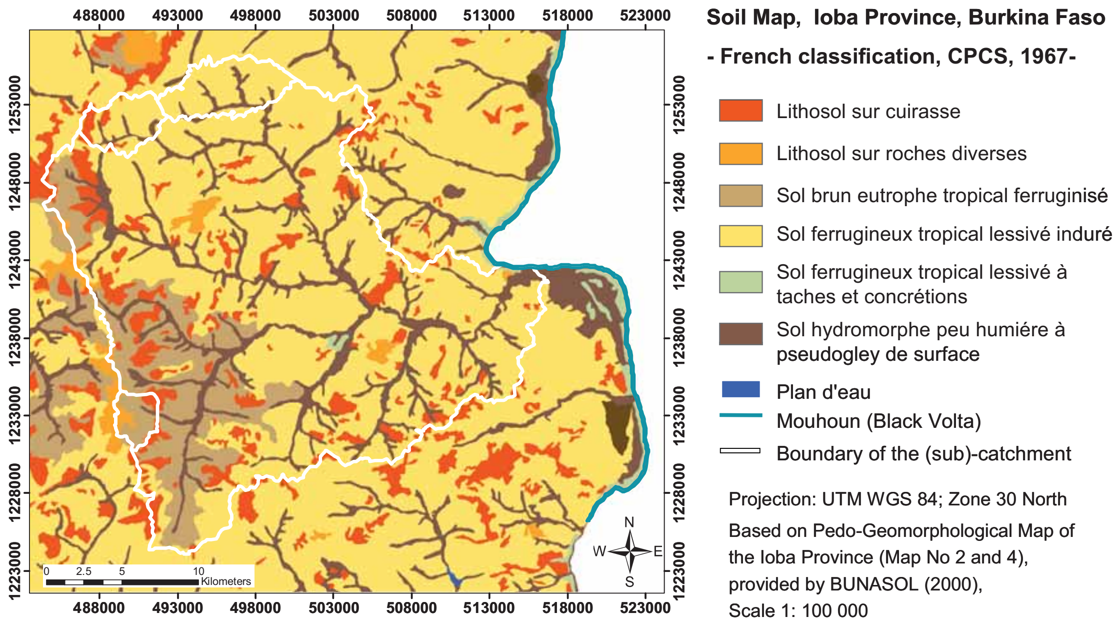


Figure 4.8 Soil map of Ioba Province, southwestern Burkina Faso

The distribution of soil types in the Ioba Province shows a predominance of *ferric lixisols* (WRB, 1998) – corresponding to *sol ferrugineux tropicaux lessivé* (CPCS, 1967) – which cover more than 60 % of the total catchment area (Table 4.5). These soils are also the dominant soil type in the sub-catchments of Wahable and Fafo, and show a medium erodibility with average K-values of 0.12. *Cambisols* (WRB, 1998), which are equivalent to *sol brun eutrophe tropical ferruginisé* (CPCS, 1967), cover only about 14 % of the total catchment area, but present the main soil type in the southwestern part of the Ioba province and thus in the sub-catchment of Dano.

Table 4.5 K-factor for dominant soil types in the Ioba Province

Soil type French classification (CPCS, 1967)	**Soil type International classification (WRB, 1998)**	**Area km² (%)**	**K-factor value**		
			Roose and De Noni (2004)	Measured (field survey)	Input value for erodibility map
			(t acre h/100 acre ft tonf in)		(kg h Mj^{-1} mm)
Lithosol sur cuirasse	Plinthosol	61 (11 %)	0.01 -0.04 (0.03) [(a)]	-	4
Lithosol sur roches diverses	Leptosol	9 (1 %)	0.01-0.2	0.20-0.22 (0.21) [(a)]	28
Sol brun eutrophe tropical ferruginisé	Cambisols	79 (14 %)	-	0.19-0.29 (0.23) [(a)]	31
Sol ferrugineux tropical lessivé induré	Ferric Lixisol	344 (60 %)	0.2-0.3	0.1-0.14 (0.12) [(a)]	16
Sol ferrugineux tropical lessivé à taches et concrétions	Stagnic Lixisol	1 (0.2 %)	0.2-0.3	0.13-0.24 (0.17) [(a)]	22
Sol hydromorphe peu humiére à pseudogley de surface	Gleysol	83 (14 %)	-	0.14-0.28 (0.21) [(a)]	27

[(a)] *most appropriate or averaged value selected and converted into metric units for soil erodibility map*

Their higher average K-value of 0.23 expresses a slightly higher susceptibility to soil particle detachment. Soils along temporarily flooded streams and flow paths – namely *Gleysols* (WRB, 1998) or *sol hydromorphe peu humiére à pseudogley de surface* (CPCS, 1967) – cover about 14 % of the total area and show similar erodibility values to those of cambisols, with some variation dependent on their texture. *Plinthosols* (WRB, 1998), which correspond to *lithosols sur cuirasse* (CPCS, 1967), are most resistant to erosion. Their average K-value of 0.03 is a result of their massive plinthic structure and the irreversible hardening of iron-rich soil particles forming a ferruginous laterite crust ("cuirasse"). All other soil types, namely *leptosols* (WRB, 1998), equivalent to *lithosol sur roches diverses* (CPCS, 1967), and *stagnic lixisols* (WRB, 1998), equivalent to *sol ferrugineux tropical lessivé à taches et concretions* (CPCS, 1967), play a minor role as they cover less than 1 % of the total catchment area.

Land-cover map and erosion potential

The crop cover factor (C-factor) determines the susceptibility of different land-use/crop-cover types to soil erosion, and was derived directly from classified land-cover maps of ASTER-images (section 4.3.7).

Table 4.6 Land-cover map of Ioba-catchment based on Aster image 2004

LUC Code	**LUC-Name** ***(IGN-FI 2005)***	**LUC-Definition**	**Area (km²)**	**Area (%)**
101	Annual crops (*cultures annuelles*)	Rainfed cultivation, mainly sorghum, millet, maize, but also cotton, groundnuts, beans and vegetables (including irrigation agriculture).	185.7	32.0
102	Burned area [(a)] (*zones incendées*)	Largely burned grass or shrubland areas (controlled bush fires), often on plateau areas, but also a common practice on fields after harvest.	7.5	1.3
103	Open woodland/ River riparian vegetation (*forêt claire/ forêt galerie*)	Tree coverage of 50-70 %, more or less joining crowns, closed canopy, occurrence often along permanent and non-permanent flows and along slopes.	92.1	15.9
104	Grassland (*savane herbeuse*)	Typical natural savannah type, grass height not exceeding 80 cm, only scattered trees, especially in dry season sparse and desiccated.	61.4	10.6
105	Shrubland (*savane arbustive/ arborée*)	Typical and predominant savannah type for the climate zone, vegetation is either dominated by shrubs (arbustive) or mixed with trees up to 50 % (arborée).	201.7	34.8
106	Settlement - artificial surface (*habitat rural - surface artificialisée*)	Village areas and intensive rural settlement with clustered houses, relatively high population density and artificial cover between 30-80 %.	3.5	0.6
107	Road network (*réseaux routiers*)	Asphalted and rural roads accessible by car.	0.5	0.1
108	Rural settlement – bare soil (*espace rural – sol nus*)	Extensive rural settlement with dispersed homesteads, small fields, common open spaces, only sparse vegetation and predominantly bare soil.	24.9	4.3
109	Dam/ water body (*barrage/ plan d´eau naturel*)	Small reservoirs and watering places (“retenue d´eau”)	1.0	0.2
110	Rivers/ perennial, seasonal flows *(Rivière/ cours d´eau permanent et temporaire)*	Flow network with first order streams (“Mouhoun” or Black Volta) and second/third order flows with only seasonal water	1.1	0.2
	TOTAL		**579.5**	**100**

[(a)] *Burned areas were later reclassified as grassland, which was the predominant permanent vegetation type for these areas.*

The land-cover map was specially adjusted to the needs of erosion hazard mapping and to the requirements of soil erosion modeling. Therefore, the land units were classified based on their surface cover and hence erosion potential into 10 classes: cultivated areas (annual crops), natural and semi-natural vegetation (open woodland/river riparian vegetation, grassland, shrubland, burned areas and rural settlement/bare soil), nonvegetated areas (river/permanent and temporary flows; dams/water bodies), and artificial areas (settlement/ artificial surface; roads).

Definitions of land-cover classes were closely linked to the nomenclature of the National Institute of Burkina Faso (IGN-FI, 2005) (Table 4.6).Based on a literature review on the land-cover values used for soil erosion modeling with special emphasis on West Africa (Table 4.7), annual crops are assigned to a C-factor of 0.4 as suggested by Morgan (1995), Mati and Veihe (2001), and Roose and De Noni (2004). The C-factor is defined as "the ratio of soil loss from land cropped under specified conditions to the corresponding loss from clean-tilled, continuous fallow." (Wischmeier and Smith, 1978, p. 17).

Table 4.7 Literature review on C-factor values

Land-cover type/ (LCC-Name, Aster 2004)		**C-factor values**				***Selected C-factor value for Burkina Faso***
		Wischmeier and Smith (1978)	Morgan (1995)	Mati and Veihe (2001) East and West Africa	Roose and De Noni (2004) West Africa	
Maize, millet, sorghum (*Annual crops*)	maize		0.2	0.43 (Aina et al, 1979)		
	sorghum/ millet	0.02-0.1	0.4-0.9	0.48 (Bonsu and Obeng, 1979)	0.4-0.9	0.4
Overgrazed or burned savannah/ prairie grass (*Grassland & burned area*)		0.1	-	0.4-1.00 (Mati, 1999, rangeland)	0.1	0.1
Forest/ dense shrub (*River riparian vegetation*)		0.001	0.001-0.002	-	0.001	0.001
Savannah/ prairie grass in good condition (*Shrubland*)		0.01	01-0.1	0.003-0.3 (Mati, 1999)	0.01	0.01
Bare soil (*Rural settlement – bare soil & artificial surface*)		1	1	-	1	1

The C-factor takes both cover and management variables into account and varies between 0, expressing that no erosion occurs (e.g., dense forest), to 1, indicating the maximum of erosion (e.g., bare soil).

The land-cover map for the Ioba-Catchment (Figure 4.9) shows that a high percentage of the area (32 %) is under annual crops, which include predominantly rainfed crops such as millet, sorghum and maize. A common practice after harvest is to expose fields, but also large plateau areas, to more or less controlled bush fires. In the land-cover map, the percentage of burned areas (1 %) is low because the season for burning had just started. These areas were later assigned to grassland, which was identified as the predominant permanent vegetation type for these areas.

The distribution of open woodland/river riparian vegetation (15 %) closely follows the course of perennial and seasonal flows, and clearly reflects the flow network of the catchment. Open woodland is also found on some sloped areas, and its coverage increases with distance to densely populated areas. The dense, mainly undisturbed vegetation is assigned a commonly used C-factor of 0.001. The highest percentage of the area is covered by the typical savannah vegetation grassland (10 %) and shrubland (34 %).

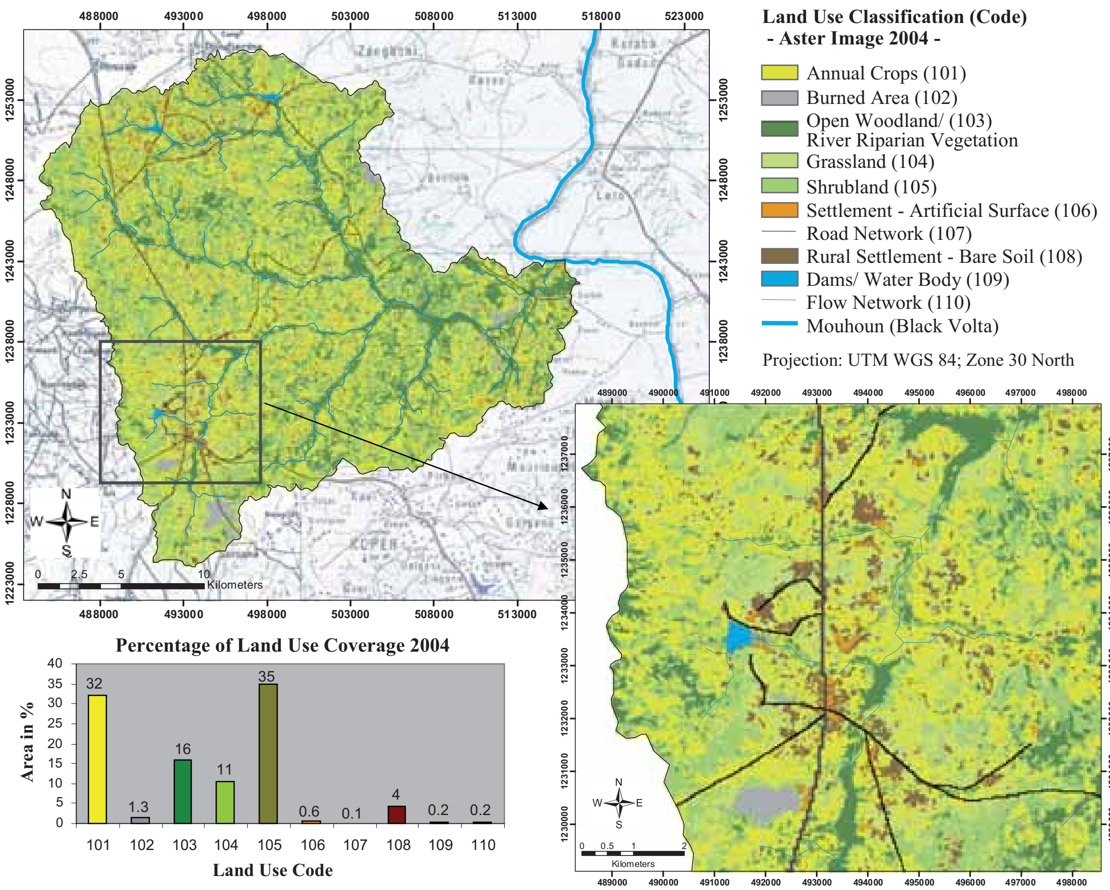


Figure 4.9 Land-cover map of the Ioba-catchment and the sub-catchment of Dano (classified Aster image 2004)

Grassland can be characterized by degraded, less fertile land, which is fragmentally found on plateau areas with shallow soils and is mainly used for grazing. Due to the sparse, often overgrazed vegetation cover, a C-factor value of 0.1 was chosen, which is also proposed by Wischmeier and Smith (1978) and confirmed by Roose and De Noni (2004) for West Africa.

In contrast, shrubland shows a denser vegetation cover and occurs either widely distributed in remote areas or in areas that are difficult to access (in general very dense shrubs often mixed with trees up to 50 %: *savane arborée*). Shrubland also appears as secondary vegetation on abandoned or fallow land (dense shrub vegetation with fewer trees: *savane arbustive*). For shrubland, a C-factor of 0.01 was used as suggested by Wischmeier and Smith (1978) and Roose and De Noni (2004).

Settlement covers only a minor part of the area (<5 %), but the population density of the province Ioba with 55-75 inhabitants per km² is relatively high (Boirard et al., 2004), and natural population growth as well as positive migration dynamics will intensify the expansion in the future. Based on the different intensities of settlement, villages with predominantly artificial surfaces are distinguished from dispersed rural homesteads. Villages are classified as intensive rural settlement if they show a centralized structure, a high building density and a high percentage of artificial surfaces (30-80 %). In contrast, extensive rural settlement is characterized by dispersed homesteads interfused by open areas and common places with no or only sparse vegetation, few and very small fields, and a low percentage of artificial surfaces (<30 %). Therefore, the predominant coverage is equated to bare soil properties. The commonly suggested C-factor value of 1 is found appropriate to reflect the properties of bare soil, but is also used for intensive rural settlement.

Linear features, such as flow and road network, covering only a minor percentage of the catchment area (0.09 % and 0.18 %, respectively), were not considered separately in the erosion modeling procedure, but instead their neighboring cell values were used (e.g., for flows mainly river riparian forest). Dams and water bodies (0.18 %) were considered as sink areas and assigned to a C-factor value of 0.001.

The WaTEM/SEDEM model offers the option to combine various land-use/land-cover/environmental feature maps into a single raster map by a GIS-overlay function. For this overlay map, the following raster files were generated: a satellite image or land-use map, representing all land-cover units as described above, an arable land map or field parcel map, differentiating single fields from each other, a forest map including the areas of "open woodland/river riparian vegetation" derived from the land-cover classification, a pasture map showing the land-cover unit "grassland" also derived from the land-cover classification, a road map considering all linear road features of at least one-pixel-width (here 15 m), a river map describing the larger stream network based on the above described SRTM-data set, and a catchment area map delineating the water catchment boundaries within the study area.

The influence of soil and water conservation measures (P-factor) was not considered, because erosion control practices were rarely applied and mostly limited to single small fields, which were often below the resolution of the raster maps.

Simulated soil erosion/deposition maps

The WaTEM/SEDEM model was run for all catchments. Sediment production, sediment deposition, sediment export, river export and pond deposition were simulated. Additionally, a soil erosion/deposition map was generated showing the redistribution rates for each raster cell of the catchment.

4.4.3 Model calibration

The model was calibrated by changing the transport capacity coefficients (Ktc) for different land-cover types in order to compare simulated results against measured sediment deposition/yield values from the bathymetrical survey (section 4.3.3). The transport capacity coefficient reflects model sensitivity to overland flow and hence sediment delivery. A lower $K_{TC\text{-}F}$ value ranging from 10 m to 100 m was given to well-vegetated areas (e.g., forest, shrubland), and a higher $K_{TC\text{-}A}$ value ranging from 30 m to 300 m was assigned to poorly vegetated surfaces (e.g., arable land, grassland and bare surfaces). For each of the three catchments, the model was run in intervals of 10 m ($K_{TC\text{-}F}$) and 30 m ($K_{TC\text{-}A}$). Sediment yield (SY) and area-specific sediment yield (SSY) were calculated and compared with measured values. The model efficiency (ME) equation by Nash and Sutcliffe (1970) was used as a measure of likelihood to select the optimal combination of K_{TC} values:

$$ME = 1 - \frac{\sum (Y_{obs} - Y_{pred})^2}{\sum (Y_{obs} - Y_{mean})^2} \tag{4.10}$$

where ME = model efficiency, Y_{obs} = observed sediment yield, Y_{pred} = predicted sediment yield, and Y_{mean} = average observed sediment yield.

The values for model efficiency can range from -∞ to 1, where values approximating 1 indicate a lower initial variance and hence a better estimate for individual sediment yield values. The most appropriate parameter combination was reached for K_{TC-F} at 80 m and for K_{TC-A} at 240 m, resulting in an optimal model efficiency of 0.79 (Figure 4.10).

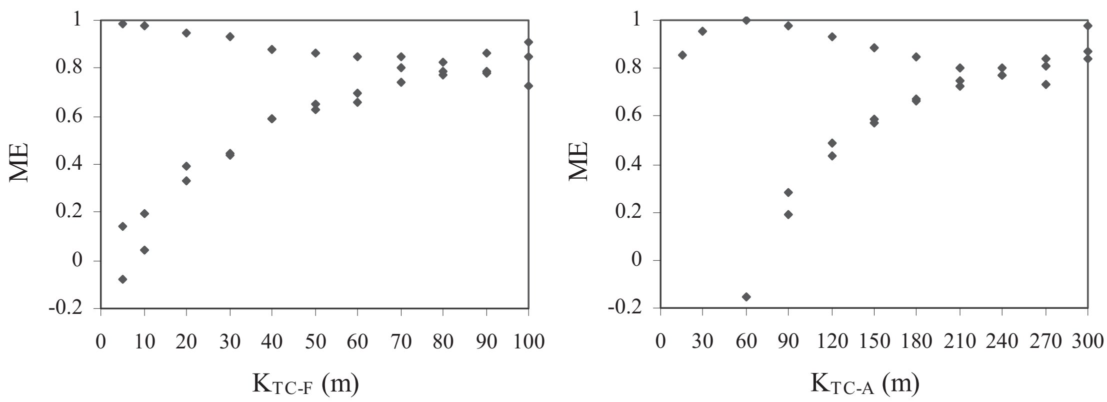


Figure 4.10 Calibration of WaTEM-SEDEM

4.5 Results and discussion

4.5.1 Initial and actual stage-storage and stage-area curves

The loss of water storage due to siltation is expressed by the stage-storage and stage-area relationship, which is derived from the initial and actual contour maps of the reservoir. As the reservoirs of Wahable and Fafo have experienced approximately 20 years of sediment inflow by water erosion since the dams were built, accumulated soil particles at the bottom of the reservoirs have influenced the reservoir morphology and led to changes in water storage capacity and surface area (Figure 4.11 and 4.12)

In 1984, the Wahable reservoir had an initial water storage capacity of 480 x 10^3 m^3 at normal pool level with an elevation of 300.4 m, a water level of 4.2 m and a surface area of 29 ha. The normal pool level is reached just before the spillway is flooded and is defined as the elevation at which water is normally stored. It presents at the same time the top of the active storage of the reservoir. The maximum pool level (also referred to as maximum flood control level) was 680 x 10^3 m^3 at a water level of 4.8 m and an area coverage of 34 ha. This maximum level is the highest acceptable water storage and water surface elevation at which the safety of the reservoir is still ensured. It can be derived from the stage-volume-area curve as the point at which the surface area curve exceeds the volume curve and presents the limit of an acceptable surface-volume relationship. Above this stage, the area requirement would be over-proportional compared to the water volume, resulting in an extensive flooding of the surrounding area (probably farmer fields) without an appropriate gain of water storage. The absolute maximum water capacity of the reservoir at which the dam crest would be flooded was calculated to be 1000 x 10^3 m^3 at a water level of 5.7 m and a corresponding surface coverage of 38 ha. The inactive storage, defined as the water volume stored below the spillway, conduit or irrigation canal, had a volume of 44 x 10^3 m^3 at a water level of 1.6 m and covered an area of 6 ha. The inactive capacity presents the minimum water surface level below which the reservoir will not fall under normal conditions. Based on the more recent bathymetric map of 2005, the stage-storage relationship changed significantly. At normal pool level, the water storage capacity was reduced by 10 % from 480 x 10^3 m^3 to 440 x 10^3 m^3, whereas the inactive storage decreased by 60 % from 44 x 10^3 m^3 to 17 x 10^3 m^3 covering 4 ha, i.e., 66 % of the initial area.

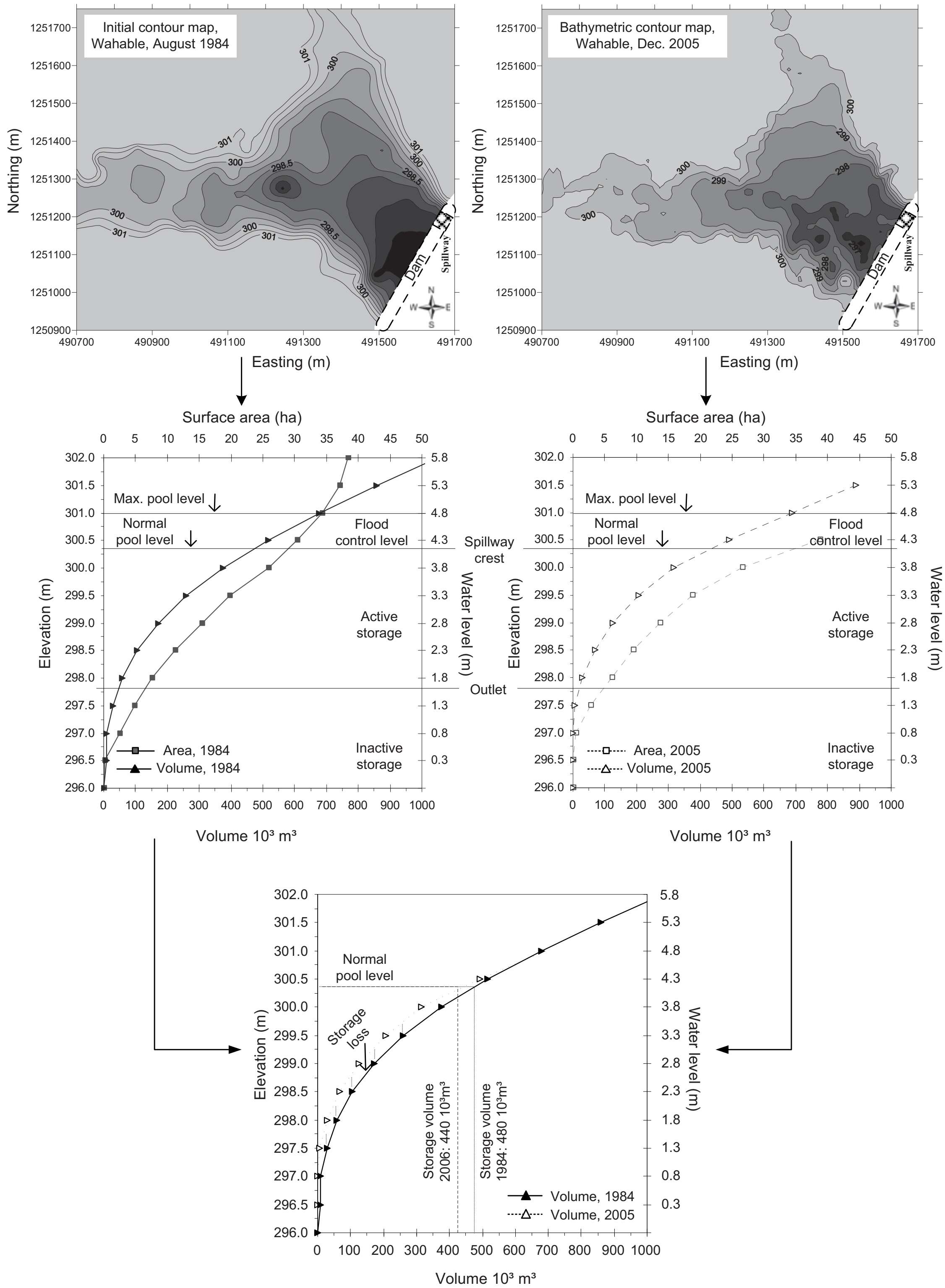


Figure 4.11 Stage-volume and stage-area curves of the Wahable reservoir in 1984 and 2005 based on the initial topographical contour map and the actual bathymetric map

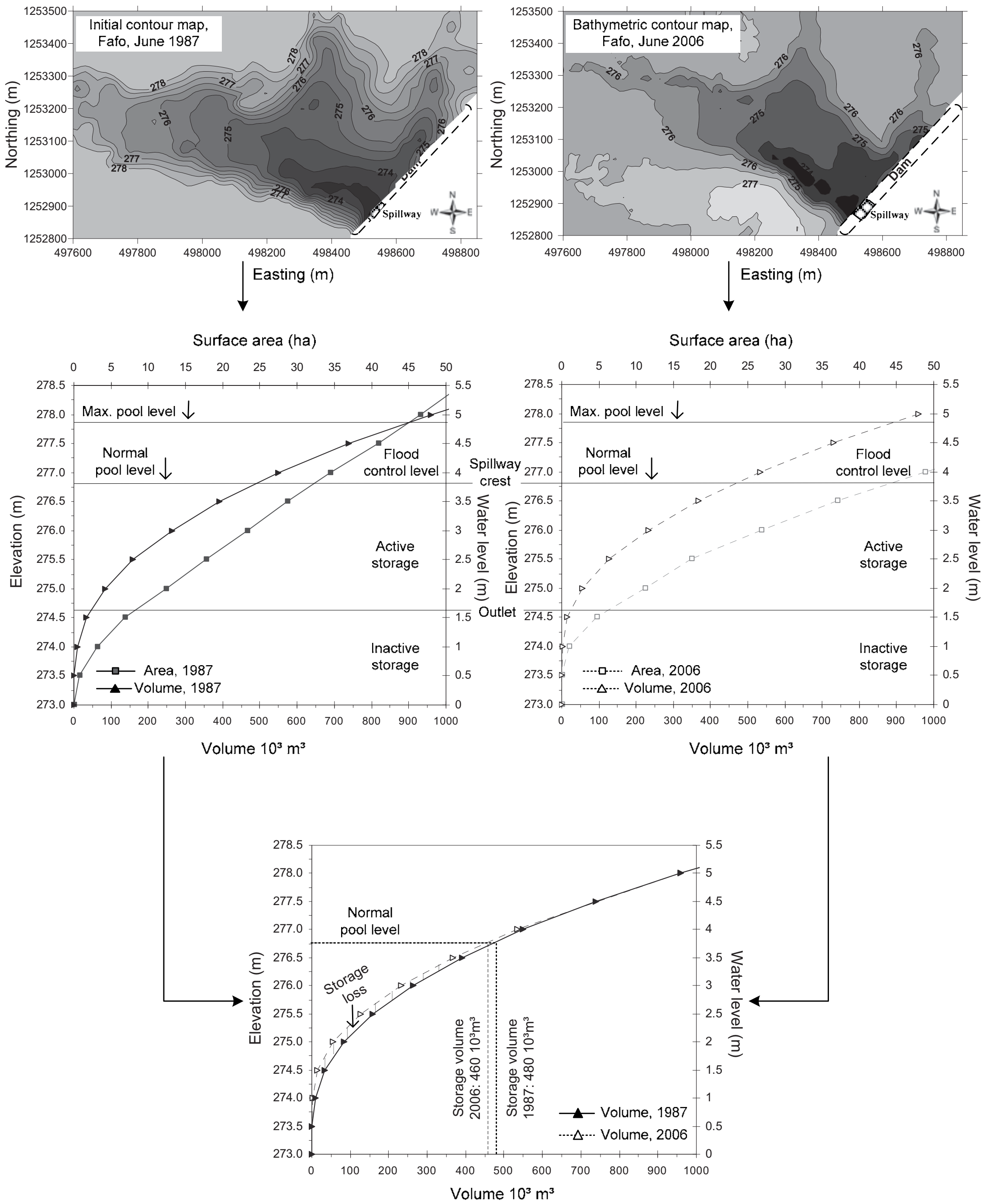


Figure 4.12 Stage-volume and stage-area curves of the Fafo reservoir in 1987 and 2006 based on the initial topographical contour map and the actual bathymetric map

The original storage capacity of the Fafo reservoir at normal and maximum pool level was of the same magnitude as that of Wahable, but the surface area coverage was higher. The large extent of the reservoir and its shallow water depth is a result of the flat terrain in the Fafo district with low slope gradients of 1-2 % and smooth shorelines. Therefore, a small increase in water level leads to a great surface area.

At normal pool level, the initial water storage capacity was 480 x 10^3 m^3 at an elevation of 276.8 m, a water level of 3.8 m and a corresponding area coverage of 32 ha (Figure 4.12). The surface area increased to 44 ha at a water level of 4.8 m when the maximum pool level with a water volume of 870 x 10^3 m^3 was reached. The absolute maximum water capacity comprised 1000 x 10^3 m^3 at a water level of 5.1 m covering an area of 48 ha. Even the inactive storage with a volume of 54 x 10^3 m^3 below a water level of 1.6 m required a surface area of 10 ha. Based on the more recent bathymetric map of 2006, at normal pool level, the water storage was reduced by approximately 4 % from 480 x 10^3 m^3 to 460 x 10^3 m^3, while the surface area increased by almost 30% to 44 ha. The inactive storage decreased by about 65 % from 54 x 10^3 m^3 to 19 x 10^3 m^3 covering 6 ha, i.e., only 60 % of the initial area.

Although the loss in water volume at normal pool level is the most common indicator for long-term changes, the reservoirs in Burkina Faso reach this level only a few weeks in a year, mostly at the peak of the rainy season in August. Therefore, it might be more meaningful to look at water storage changes in relation to water level heights (Table 4.8). In general, the lower the water level, the higher the loss of storage. The loss of water storage in Wahable reaches between 16 % and 50 % at water levels ranging from 3.3 m to 1.8 m. In Fafo, the storage loss shows a similar amplitude ranging from 12 % to 88 % at water levels from 3 m to 1 m.

Table 4.8 Loss of storage volume as function of water level

Wahable reservoir			
Water level (m)	-1984- Storage volume (10^3 m^3)	-2005- Storage volume (10^3 m^3)	Storage loss (%)
3.8	375.7	315.8	15.9
3.3	260.0	205.9	20.8
2.8	172.8	126.0	27.1
2.3	105.9	68.4	35.5
1.8	58.9	29.3	50.4

Fafo reservoir			
Water level (m)	-1987- Storage volume (10^3 m^3)	-2006- Storage volume (10^3 m^3)	Storage loss (%)
3.0	263.0	232.5	11.6
2.5	159.4	126.6	20.6
2.0	84.4	54.2	35.8
1.5	34.4	14.5	58.0
1.0	10.1	1.2	88.0

However, at lower water levels the reservoir has a comparatively higher and at higher water levels a comparatively lower storage loss compared to Wahable due to the flatter terrain. The shallow water depth causes a faster sedimentation of the inactive storage, and consequently leads to a higher spatial extension and hence a higher risk of flooding.

Both reservoirs have already lost more than 50 % of the storage capacity for water levels below 2 m. A 2-m water level is a typical threshold value for the dry season and the onset of the rainy season. Considering their original capacity at normal pool level, the reservoirs have lost about 10 % to 15 % of their capacity, and might reach their half-life in about 20-25 years. The concept of half-life expresses the time required to fill in 50 % of the original capacity, and is an indicator of the time frame in which sedimentation may seriously affect water supply and flood control. However, it does not indicate the additional time it will take to reach complete sedimentation because sediment trapping declines with reduced storage capacity (Morris and Fan, 1997).

In contrast to the reservoirs of Wahable and Fafo, the reservoir of Dano is an example of a recently built reservoir (construction 2001/2002, initial survey 1999) and had experienced only five years of sedimentation by the time of sampling (beginning of 2006). This time span is seen as too short to allow an adequate assessment of changes in contour lines, and therefore the site could not be considered for further analyses. Nevertheless, the initial stage-volume curve is briefly presented and discussed here. In 1999, the Dano reservoir had an initial water storage capacity of 360 x 10^3 m^3 at normal pool level with an elevation of 305.7 m, a water level of 5.7 m and a surface area of 16 ha (Figure 4.13). The maximum pool level is 720 x 10^3 m^3 at a water level of 7.5 m and an area coverage of 24 ha. The absolute maximum water capacity of the reservoir was calculated to be 1050 x 10^3 m^3 at an absolute maximum water level of 8.7 m and a corresponding surface coverage of 30 ha. The dead or inactive storage had a volume of 45 x 10^3 m^3 at a water level of 2.7 m and covered an area of 4.5 ha.

The reservoirs in Wahable and Fafo are losing their capacity at annual rates of 0.4 % and 0.2 %, respectively, which corresponds well with other studies from northern and western Africa. For 16 dams in Morocco, Lahlou (1993) showed that the average annual loss rate was 0.57 %, ranging from 0.11 % to 1.6 %. For the half-life of the dams, he calculated a capacity loss of 47 % in 30 years for the Oued Nakhla dam and of 16 % in 10 years for the Oued Nekor dam in Morocco.

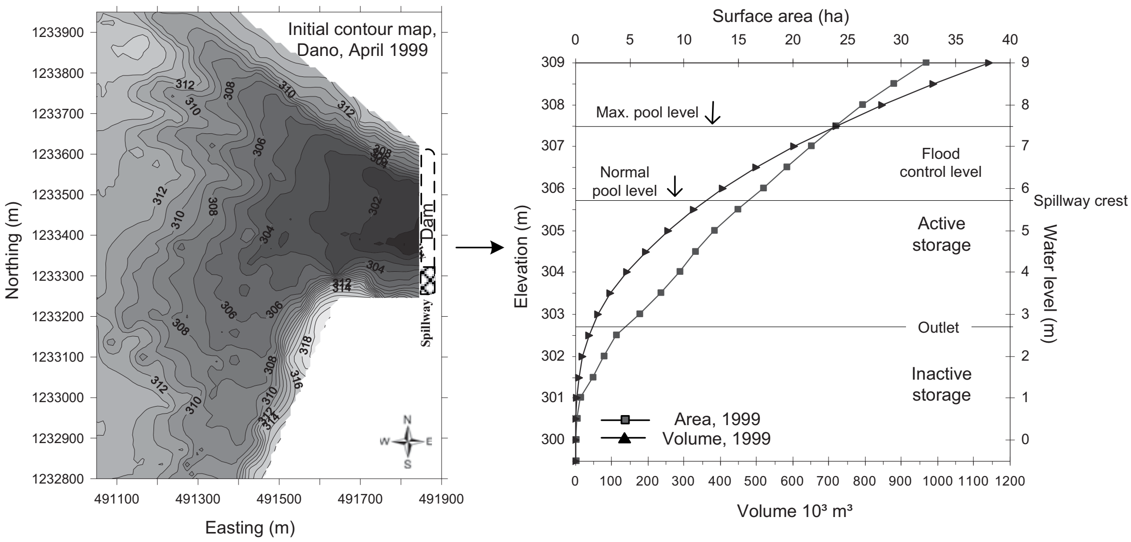


Figure 4.13 Stage-volume and stage-area curves of the Dano reservoir in 1999 based on the initial topographical contour map

Albergel et al. (2004) estimated an average loss in storage capacity of 22 % in 8 years for more than 20 small dams in Tunisia, which would be equivalent to an average loss rate of 2 % to 3 % per year. Such high sedimentation rates and short life expectancies were also reported by Fox et al. (1997). As the slope gradients in Burkina Faso are lower, sedimentation rates are correspondingly lower than the depletion rates of these steeper mountain reservoirs, but even on the gentle topography of Burkina Faso some significant changes in storage loss can be recorded.

Generally, it is known that smaller reservoirs show a relatively faster loss in storage volume (Morris and Fan, 1997). While larger dams have annual loss rates between 0.2 % and 0.8 %, which corresponds to a half life of 91 to more than 400 years, small dams, in contrast, with initial reservoir capacities of less than 1.23 Mm^3, can have annual depletion rates between 1 % and 3.5 %. This would corresponds to a half life of 25 to 81 years. This inverse relationship between pool volume and sedimentation rate was ascertained by Dendy et al. (1973) by examining more than 1000 reservoirs in the USA.

However, to-date, only a few studies exist on siltation rates of small dams in the Volta Basin. Due to a lack of basic hydrological and sedimentological data, most of the studies focus on the collection and elaboration of detailed data about number, size, pool volume and specific properties of small reservoirs (Cecchi, 2009), while other studies aim to develop appropriate methods to capture such data and to detect spatial and temporal changes by remote sensing techniques (Liebe et al., 2005). Once this knowledge is acquired, further research should focus on the severity of siltation and its impact on storage loss and life expectancies for small dams in Burkina Faso. The high number of small dams below 1000 x 10³ m³ (Figure 4.2) in Burkina Faso, and the dependency of farmers on this water for domestic use, livestock and irrigation, underlines the need for a proper assessment of their life span through siltation studies.

4.5.2 Calculation of sediment thickness along a longitudinal transect

Changes in reservoir bed morphology and loss of storage volume along a reservoir transect is a useful direct indicator of sediment thickness. The accretion of accumulated sediment is visualized by comparing the morphology at the bottom of the reservoir between the initial topographical map and the recent bathymetrical map. The difference in morphology is presented along a longitudinal cross section for the reservoirs of Wahable and Fafo and shows in both cases a significant increase in sediment accumulated during the past approximately 20 years (Figure 4.14 and 4.15).

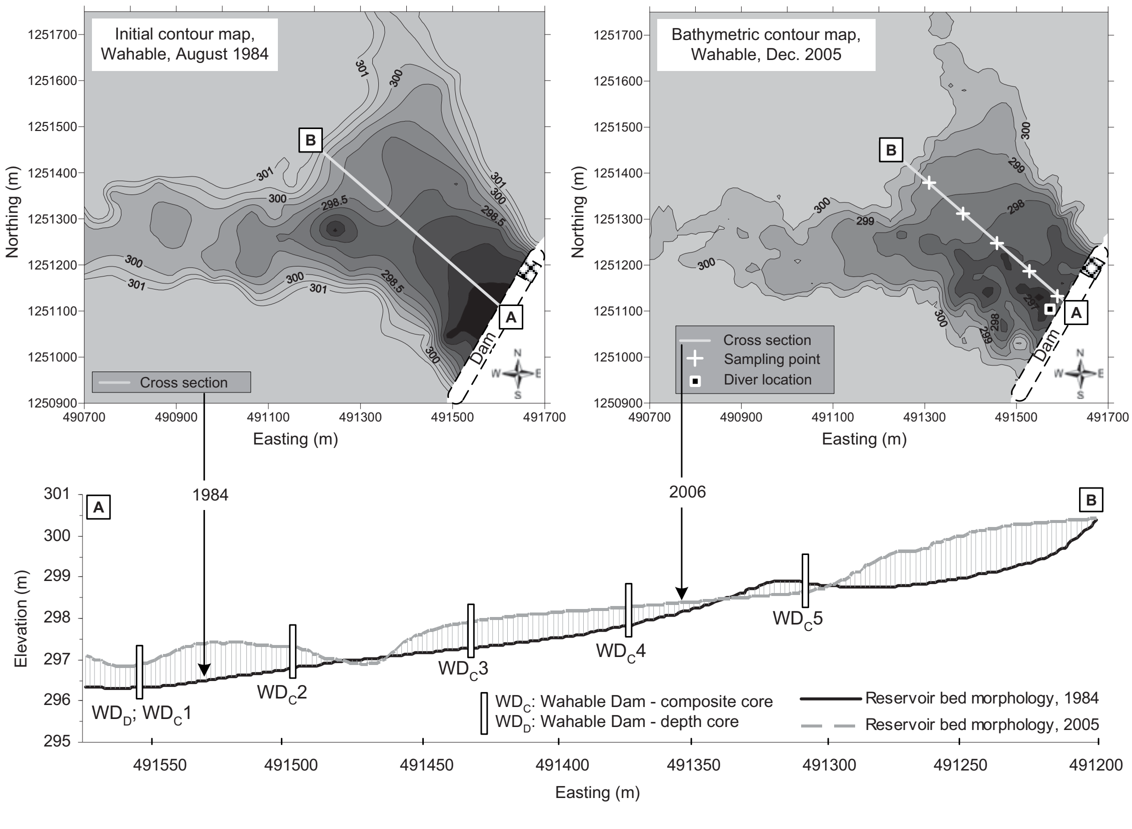


Figure 4.14 Cross section of reservoir bed morphology at two time points (1984 and 2006), Wahable

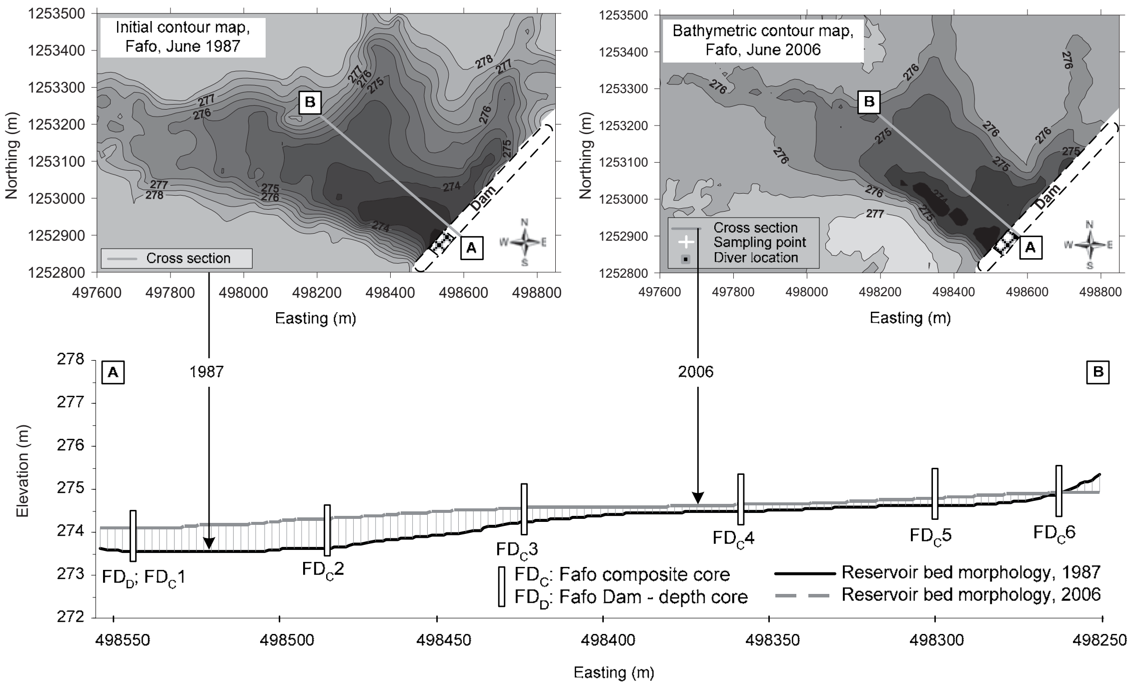


Figure 4.15 Cross section of reservoirs bed morphology at two time points (1987 and 2006), Fafo

Whereas the average sedimentation thickness in Wahable is 0.5 m and reaches maximum of 1 m, the average thickness in Fafo is 0.3 m and does not exceed more than 0.7 m at any location along the cross section. Compared to the almost smooth accumulation line of Fafo, which continuously decreases from the dam (A) to the shallow shorelines (B), the accumulation line in Wahable fluctuates.

At some locations, the actual accumulation line of Wahable falls even below the initial accumulation line, probably due to continuous erosion along former erosion rills or actual recess along inflow channels. Another reason for the distinct variations in sedimentation pattern might be the high number of remaining trees in the reservoir influencing stream inflow as well as the settling characteristics of sediment particles near such trees. The relatively high sediment thickness at the shoreline (B) in Wahable could be a result of the deeper water level and the more abrupt increase of lateritic shorelines bordering the reservoir area. Due to the more distinct basin character of the reservoir in Wahable, the water level is generally 1-2 m deeper than that in Fafo, and most of the reservoir area is permanently filled with rain and/or runoff water. The near-level shallow reservoir basin in Fafo, in contrast, causes low water levels leading either to extensive seasonal flooding of the surrounding areas during the rainy season or to a drying-up of larger reservoir areas during the dry season. In the dry season, the shoreline area is permanently disturbed due to animal watering/ grazing and to human activities, leading to a removal or redistribution of sediments. This is also reflected in the shape of the initial accumulation line, which exceeds the actual accumulation line. The generally higher sedimentation thickness in Wahable might be explained by the more undulating terrain with 1-2° higher slope gradients than in Fafo, even though these differences in terrain attributes are relatively small. Human-induced processes such as changes in land management or accelerated or exceptional deforestation are not reported for any of these villages since 1985. However, the region is experiencing a steady demand for additional agricultural land and a permanent need for firewood.

Another reason for higher sediment accumulation in Wahable could be the smaller size of the contributing area, which might lead to a comparatively faster sediment transport through the catchment and hence to a faster deposition at the dammed outlet. The size of the catchment is with 14.8 km² approximately 30 % smaller than the catchment area of Fafo (23.6 km²), which corresponds with the higher accumulation thickness in Wahable. An inverse correlation between catchment size and sediment yield has been established (A.S.C.E., 1975; Walling, 1983), which is mainly related to decreasing slope and channel gradients with increasing catchment size; this increases deposition of sediment before it reaches the basin outlet. Many studies indicate that often less than 30 % (e.g., Walling et al., 2001) and in some cases less than 5 % (e.g., Walling and Collins, 2008) of the gross soil loss is transported to the outlet of the catchment. The larger part of the mobilized sediment is deposited in intermediate storages, such as sink areas, depressions, or ephemeral channels and other pathways in the drainage basin. However, Walling (1983) also emphasized that this relationship is not generally applicable to all reservoirs due to the complexity of sediment delivery processes. This is confirmed by more recent studies analyzing the impact of various environmental factors on sediment yield (de Vente and Poesen, 2005; Foster et al., 2007; Haregeweyn et al., 2008; Tamene et al., 2005; Verstraeten and Poesen, 2001). Avendaño Salas et al. (1997) show that the catchment area alone explains only 17 % of the variability in sediment yield by analyzing 60 catchments in Spain. Verstraeten and Poesen (2003) studied more than 20 of these catchments in order to determine the remaining variability in area-specific sediment yield by considering climate, topography and land-use characteristics. By using a classical multiple regression approach and a qualitative factorial scoring index they found that none of the individual parameters alone could explain the observed variability in area-specific sediment yield. In fact, the factorial scoring index showed that about 80 % of the variability might depend on the interaction of these variables. In Burkina Faso, climate, topography and land-use characteristics are very similar, so that a relationship might be observed between catchment size and sedimentation rates. However, further research is needed to explain the driving variables triggering the sediment delivery process as well as to identify spatial dynamics, sediment sources, pathways and sinks within the complex drainage system.

4.5.3 Quantification of accumulated sediment by soil core analysis

The retrieval of sediment cores from the bottom of the reservoirs allows quantification of the magnitude of the accumulated soil layer by analyzing changes in chemical and physical soil properties with increasing depth.

Both reference depth profiles of the reservoirs in Wahable and Fafo show clear down-core variations in soil properties and indicate significant stratigraphic changes at the depth where the initial reservoir bed is reached (Figure 4.16 and 4.17).

For the Wahable reservoir, an abrupt change in soil texture is visible at a depth of approximately 40 cm where the texture alters from clay (>60 % clay; >30 % silt) to loam (<30% clay; <35 % silt). The high percentage of fine clay in the upper layer is explained by the high portion of soil particles transported into the dam by incoming water from main tributaries and runoff. Additionally, clay particles might be carried by wind erosion into the dam, especially by the Harmattan at the end of the dry season. Whereas silty soil particles of the finer fraction (below 0.02 mm) can be easily transported into the reservoir by streams and heavy rainfall, sandy soil particles with a coarse fraction size (above 0.63 mm) are generally too heavy to be transported by water or runoff over large distances in flat environments. Thus, they are not found in the accumulated upper layer of the reservoir bed. The in-situ reservoir soil, however, can be recognized by its texture change to a sandy loam to loam, which is similar to the sandy soils on the hillslopes in Wahable.

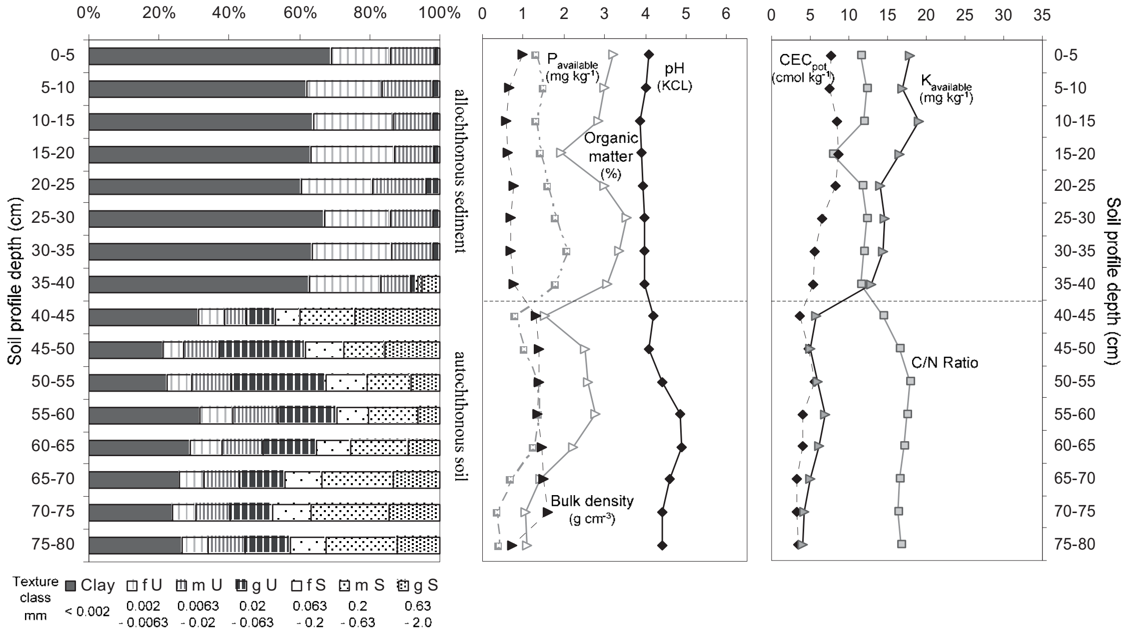


Figure 4.16 Sediment core analysis of the reservoir in Wahable (WDD): Vertical changes in soil texture and chemical soil properties with increasing depth

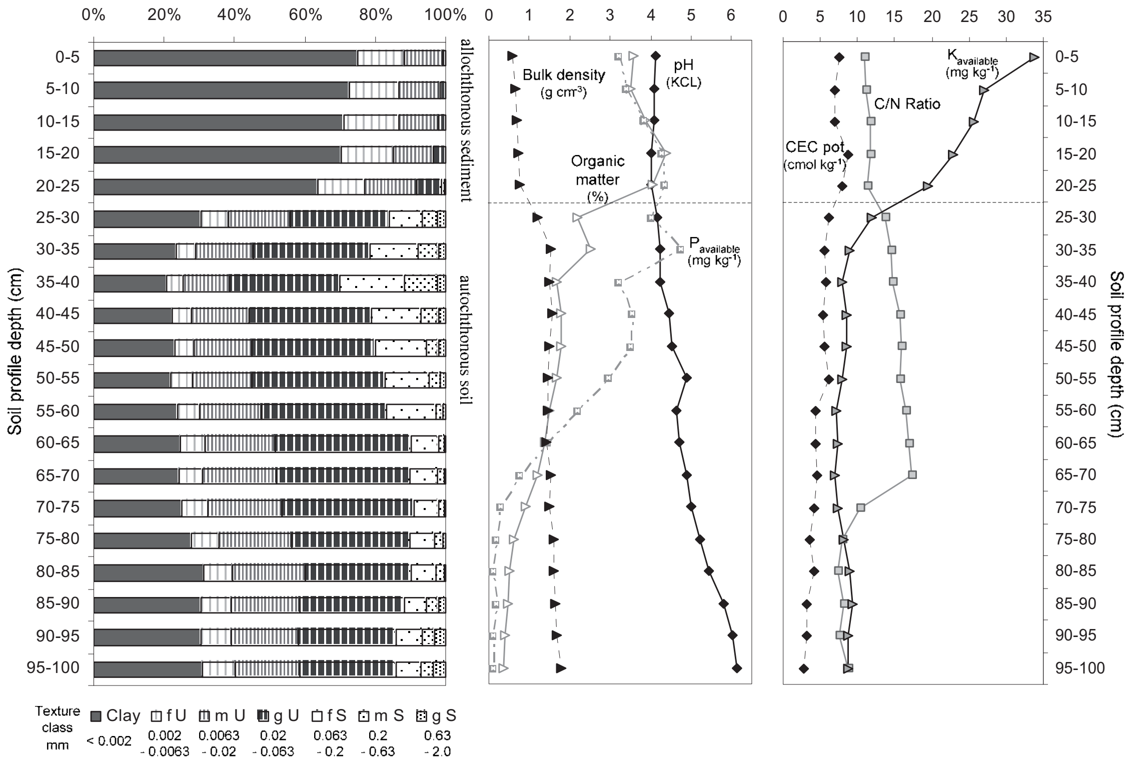


Figure 4.17 Sediment/soil core analysis of the reservoir in Fafo (FDD): Vertical changes in soil texture and chemical soil properties with increasing depth

Compared to the high percentage of approximately 60 % sand and 30 % silt for hillslope soils in Wahable (section 3.5.2), a texture of approximately 40 % sand and 30 % silt is found in the in-situ reservoir bed soils. The decrease in the coarse sand fraction and the higher portion of clay in the in-situ soils is mainly related to the landscape position in a valley, which has been a permanent sink area for accumulated material from upslope.

The clear change in other soil properties is also an evidence of an allochthonous sediment layer above an autochthonous soil. Due to the deposition of carbon-rich clay particles, the organic matter content is higher in the accumulated sediment layer (>3 %) than in the in-situ soil profile (<2 %) and shows a continuous decrease with depth, especially in the former B-horizon at about 60 to 65 cm depth. Soil bulk density is lower for the less compacted, accumulated sediments (average 0.7 g cm^3) of the upper layer than for the more compacted in-situ soil (average 1.3 g cm^3). Significant changes occur also in the C/N ratio from 11.4 to 16.6 at a depth of 40 cm.

As the C/N ratio is directly related to the organic matter content, down-core variations are similar, and both parameters show a sudden decline at 20 cm depth. The decline at this depth can be explained neither by an individual event nor by other soil properties. Soil available P and K values, which are known as indirect indicators for deposition, decrease significantly at a depth of 40 cm from 1.6 to 0.9 mg kg^{-1} and from 15.8 to 5.4 mg kg^{-1} respectively. The pH value has an almost stable value of 4 pH in the upper sediment layer, but increases to a value of 5 pH in the former A-horizon. The cation exchange capacity (CEC) values continuously decrease with increasing depth from values of above 7 cmol kg^{-1} to values of less than 5 cmol kg^{-1} below a depth of 40 cm.

The soil core analysis for the Fafo reservoir shows similar stratigraphic changes but approximately at 25 cm (Figure 4.17). The soil texture changes here from clay (>70 % clay; >20 % silt) to silt loam (<30% clay; >55 % silt). The high percentage of clay in the upper accumulated layer is again attributed to the transport of fine soil particles from the surrounding catchment into the reservoir. The lower fraction of sand particles can be explained by the generally more silty soils in Fafo compared to the sandy soils in Wahable.

The abrupt change at 25 cm depth is also clearly reflected in physical and chemical sediment/soil properties, especially in soil organic matter, bulk density and soil available P and K. Most of the properties fall within the range for Wahable, but the values for soil available P and K are almost twice as high in the upper accumulated layer. These high values, exceeding 4 mg kg^{-1} for $P_{available}$ and reaching up to 34 mg kg^{-1} for $K_{available}$, could be a result of the high use of NPK fertilizer on the cotton fields around Fafo. The absolute peak in $P_{available}$ at a depth of 35 cm corresponds to a relative peak in organic matter, suggesting P accumulation in the former vegetation. Values of $K_{available}$ decrease with depth and seem more related to the mineral fraction. The former in-situ B-horizon is reached at a depth of approximately 70 to 75 cm, indicated by an abrupt decrease in the C/N ratio.

These down-core variations in soil properties give clear evidence about the point at which intensive reservoir sedimentation begins, i.e., at a depth of approximately 40 cm for the Wahable reservoir and at 25 cm for the Fafo reservoir. The greater accumulation depth in Wahable is a result of both the age of the dam and also the smaller catchment size involving shorter particle traveling distances and therefore a higher sedimentation rate as discussed earlier (section 4.5.2).

Based on the calculations of the bathymetric survey (Figure 4.14 and 4.15), the sediment thickness in Wahable at the depth-core sampling point (WD_D) along the longitudinal reservoir transects reaches 0.52 cm, and in Fafo at the depth-core sampling point (FD_D) 0.56 cm. Whereas the bathymetrical calculations in Wahable correspond reasonably well with the depth-core estimates of approximately 40 cm thickness, the bathymetrical calculations for Fafo are twice as high as the depth-core estimates of approximately 25 cm. This difference in sediment thickness could be caused by the uncertainties and measurement errors inherent to the individual approaches. The bathymetric survey might have the higher potential for error propagation because the accuracy of the calculation is highly dependent on the precision of the two elevation maps, namely the initial topographical and the actual bathymetrical map.

For the bathymetric survey, the accuracy of depth measurements is limited to centimeter accuracy, rather than to the millimeter accuracy for the topographical survey, due to the limited precision of the underwater equipment (here depth sounder and sampling rod). The exact determination of height and position of the reservoir bed is hampered by the muddy water and the mobile GPS unit attached to the moving boat. Furthermore, the initial topographical map contains only relative elevation data. Thus, the height adjustment of both maps requires an independently fixed reference point outside the reservoir that has not been exposed to sedimentation or erosion. Geodetic points or fixed benchmarks are often difficult to find in rural areas or they might not be clearly detectable on the initial topographical reservoir map. Therefore, spatial or altimetric uncertainties may arise when overlaying both maps.

In addition, the retrieval of sediment cores also has fundamental sources of errors, such as measurement impreciseness that might occur due to an inaccurate vertical penetration of the core sampler. Also, the division of the soil sample into 5 cm depth increments can introduce inaccuracies in the bathymetric estimates.

For a better comparison, more depth-core samples are necessary, which was not feasible in this study due to time restrictions. However, in order to validate and verify the available results, the radionuclide approach was used as an independent supplementary method to identify variation in ^{137}Cs concentration with depth.

4.5.4 Depth distribution of ^{137}Cs reservoir sample

The reservoir sediment profiles show clear down-core variations in ^{137}Cs concentration (Bq kg^{-1}) and ^{137}Cs inventory or areal activity (Bq m^{-2}) indicating a significant change at the depth where the initial reservoir bed is reached (Figure 4.18 and 4.19).

A clear shift in ^{137}Cs distribution occurs at a depth of 40 cm for the Wahable reservoir with the ^{137}Cs concentration declining abruptly from 7.4 to 2.7 Bq kg^{-1} and the areal activity decreasing from 263 to 165 Bq m^{-2}. More than 80 % of the total ^{137}Cs concentration and almost 70 % of the total areal activity are found in the upper accumulated layer. Beside the decline in absolute ^{137}Cs values, the pattern of ^{137}Cs indicates also a fundamental change with depth. Whereas the variation in ^{137}Cs concentration and areal activity is relatively low in the upper 40 cm of the profile with an average coefficient of variation (CV) of 8.6 % and 16.8 % respectively, it increases strongly below a depth of 40 cm to a CV of 85 % and 84 %, respectively. This change is associated with the typical ^{137}Cs depth distribution pattern of a cultivated soil with an A-B horizon sequence, and shows that the in-situ reservoir bed is reached. Assuming an initial A horizon at a depth of 40 cm, the relative ^{137}Cs peak is found about 10 cm below the former topsoil, here at a depth of 50 to 55 cm.

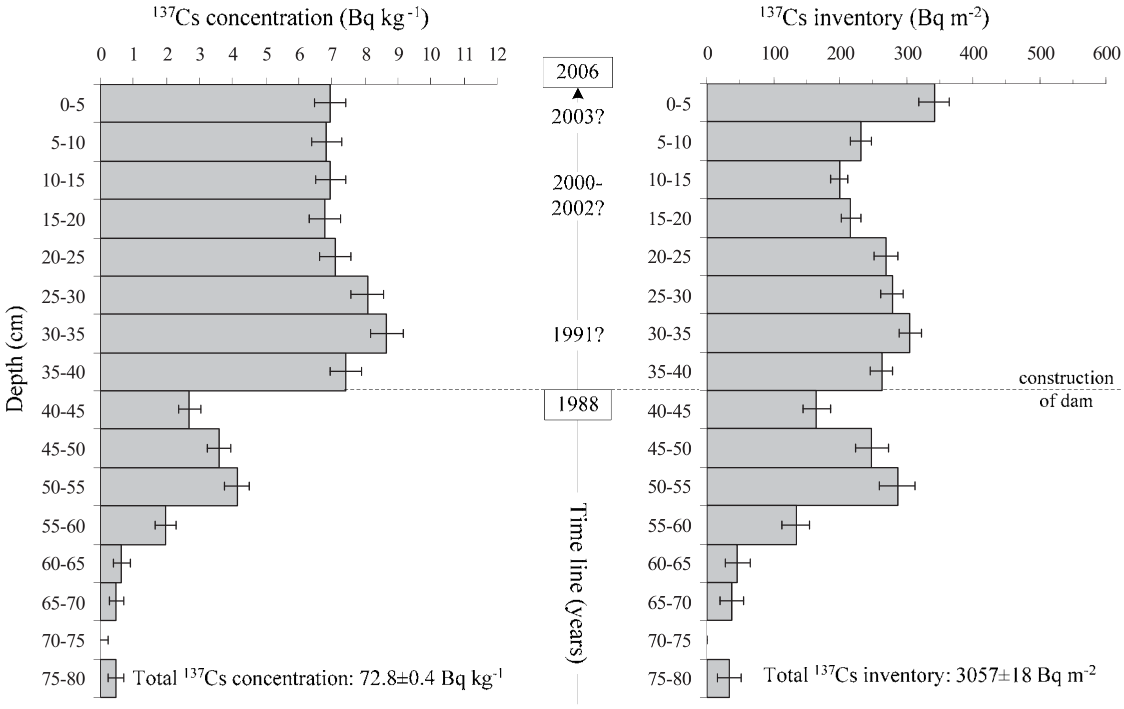


Figure 4.18 Depth distribution of ^{137}Cs concentration and ^{137}Cs inventory in a sediment core from the Wahable reservoir (WDD)

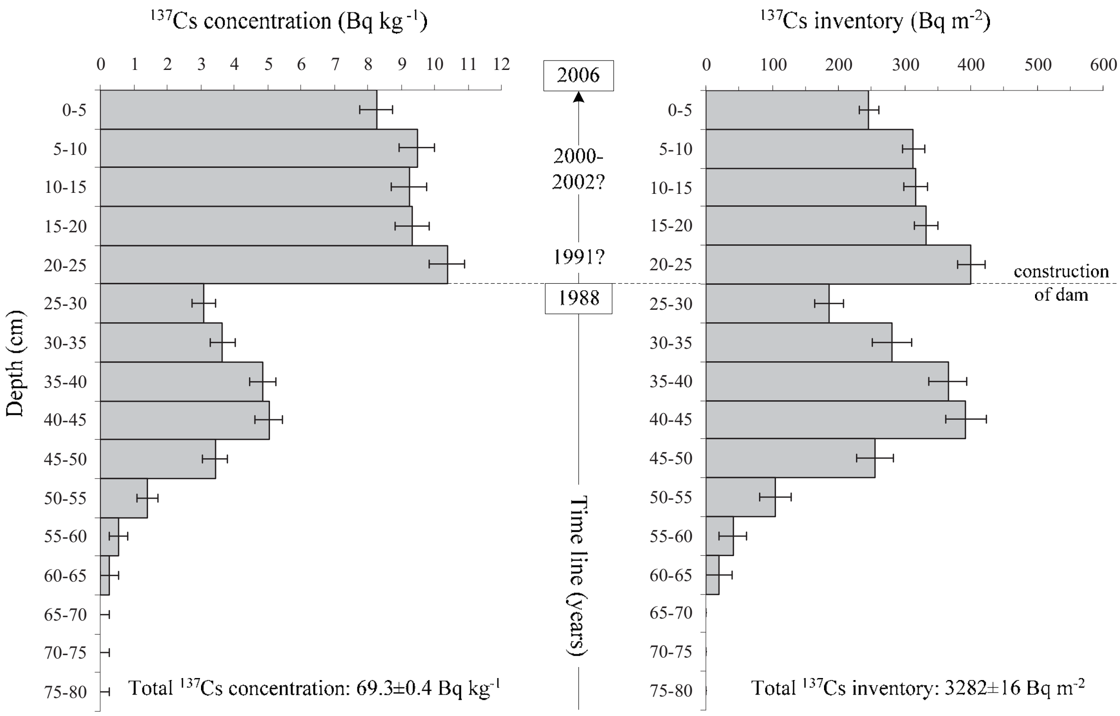


Figure 4.19 Depth distribution of ^{137}Cs concentration and ^{137}Cs inventory in a sediment core from the Fafo reservoir (FDD)

This downward displacement of ^{137}Cs could be caused by former land-use and management practices or by the vertical migration of the ^{137}Cs within the soil profile.

The ^{137}Cs distribution for the Fafo reservoir shows a similar pattern but a shallower accumulation layer, indicating that the initial reservoir bed is already reached at a depth of 25 cm. The ^{137}Cs concentration declines at this depth abruptly from 10.4 to 3.1 Bq kg^{-1}, and the areal activity decreases from 400 to 187 Bq m^{-2}. Due to the lesser thickness of the accumulation layer, the percentage of total ^{137}Cs concentration and total areal activity in the upper layer is less pronounced with 67 % and 49 %, respectively. The relatively low CV of 7 % for the ^{137}Cs concentration and 15 % for the total areal activity show again the characteristics of accumulated sediment. Below a depth of 25 cm, the CV increases to more than 95 % for both values, and the distribution pattern again shows the typical ^{137}Cs depth profile of an autochthonous soil. Also, the downward displacement of the relative ^{137}Cs peak is visible, here to a depth of 40 to 45 cm.

A comparison of the ^{137}Cs depth profiles of both reservoirs shows that the total ^{137}Cs concentration and total ^{137}Cs inventory is within the same range at both Wahable and Fafo, and that the down-core distribution has the same pattern. This confirms the suitability of the ^{137}Cs approach in differentiating the accumulated allochthonous sediment layer from the in-situ autochthonous soil profile.The results of the ^{137}Cs approach correspond closely with the sediment core results of the sediment core analyses (section 4.5.3). Both approaches indicate a significant change at the same depth intervals and can be used as complementary methods to reconstruct the maximum sedimentation depth of the reservoirs. Based on these results, an average annual gain in accumulated sediment of 2.2 cm yr^{-1} thickness is estimated for the Wahable reservoir and of 1.4 cm yr^{-1} for the Fafo reservoir .

4.5.5 Sediment production, pond deposition and sediment yield simulated by WaTEM/SEDEM

The results of the simulation by WaTEM/SEDEM provide information about the total annual amount of sediment production, sediment export, pond deposition and (specific) sediment yield at catchment scale (Table 4.9).

In the comparison of total sediment production and delivery for the three catchments, the Dano catchment shows the highest values for all simulated parameters. Annual sediment production, which is defined as the sum of net soil loss for the entire catchment, reaches almost 7600 t yr^{-1} at Dano and is approximately 5 times higher than for the Wahable catchment and 4 times higher than for Fafo. Annual sediment deposition, representing the sum of net soil accumulation within the entire catchment (not including pond deposition), is with almost 4900 t yr^{-1} in Dano almost 9 times higher than in Wahable and more than 7 times higher than in Fafo. These distinct differences can be attributed to the particular terrain characteristics of Dano such as higher slope gradients and higher relief energy leading to higher soil particle detachment, faster sediment delivery and hence higher sediment accumulation rates.

Table 4.9 Simulated parameters by WaTEM/SEDEM

Specific simulated parameter	Dano reservoir	Wahable reservoir	Fafo reservoir
Sediment production (t yr^{-1})	7594	1447	1772
Sediment deposition (excluding ponds) (t yr^{-1})	4886	552	669
Pond (=reservoir) deposition (t yr^{-1})	2409	733	847
Sediment export by river/flow/channels and other options out of the reservoir (t yr^{-1})	299	162	255
Sediment export only by river/flow/channels (t yr^{-1})	238	110	199
Sediment yield (t yr^{-1})	2708	895	1102
Specific sediment yield (t ha^{-1} yr^{-1})	3.4	0.6	0.5

The ratio of sediment production to sediment deposition is with 3:2 less pronounced in Dano compared to 5:2 in Wahable and Fafo, suggesting a comparably higher deposition rate in Dano within the landscape. This high sediment deposition rate results in a high percentage of the mobilized sediment (here about 64 %) being accumulated in intermediate storages, sinks, depressions, ephemeral channels or other pathways, which does not reach the main reservoir/pond at the outlet of the catchment. This can be ascribed to the abrupt change of slope gradients and hence the clear shift in the sediment delivery process from a high transport capacity at the steeper hillslopes of the Ioba Mountains to a low capacity on the almost flat terrain in the expanded valley, where the sediment flux exceeds the transport capacity and sediment deposition occurs.

The terrain of Wahable and Fafo has lower slope gradients of 2 to 3 % and is more suited for soil accumulation. Abrupt terrain changes are missing, and thus less than 40 % of the mobilized sediment is accumulated in intermediate storages. However, the WaTEM/SEDEM model does not consider channel deposition (nor channel erosion) and assumes that all sediment reaching a channel is transported further downslope to the outlet of the catchment, which does not reflect real conditions.

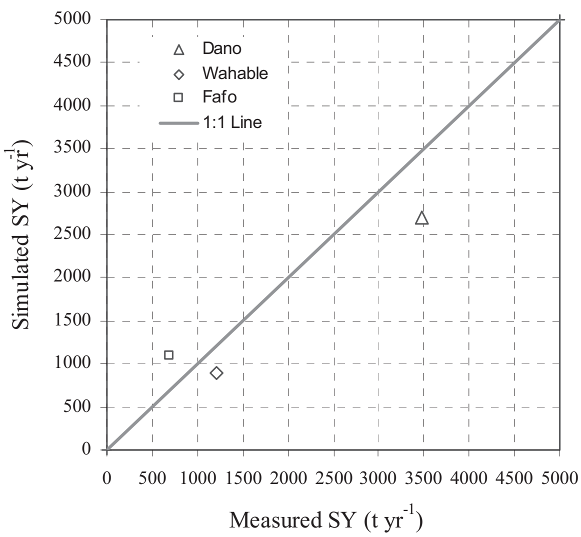
The total amount of sediment accumulated in the reservoirs, here called pond deposition, reaches 2400 t yr^{-1} in Dano and is more than 3 times higher than in Wahable (730 t yr^{-1}) and almost 3 times higher than in Fafo (850 t yr^{-1}). For model simulation, all reservoirs are accounted for by the pond option and thus the total amount of soil leaving the study area, the so-called total sediment export, represents only the sediment exported from ponds. This total sediment export value varies between 300 t yr^{-1} in Dano and 160 t yr^{-1} in Wahable and reaches up to 30 % of the pond deposition value in Fafo. About 70 % to 80 % of the total sediment is exported through rivers, or flows or channels out of the catchment. However, dams are generally less permeable than ponds and might retain sediments better, so that the amount of total sediment export might be added to the pond deposition value. Nevertheless, the exact amount is difficult to determine, as suspended sediment is lost firstly when the spillway of the dam is temporarily flooded at the end of the rainy season, and secondly when the irrigation channel is opened for agriculture at the end of the dry season.

Sediment yield considers both pond deposition and sediment export. It reflects the quantity of sediment transported by a stream or river to a specific point of the catchment, here the dam or the outlet. Similar to the value for pond deposition, sediment yield is with 2708 t yr^{-1} in Dano more than 3 times higher than in Wahable and 2.5 times higher than in Fafo. Taking the small catchment size of Dano into account, these differences become even more distinct. A comparison of gained sediment yield per km^{-2}, so-called specific sediment yield, shows that Dano produces 3.4 t $ha^{-1}yr^{-1}$, or about 5 to 7 times more sediment yield than Wahable and Fafo, respectively. In contrast, the low specific sediment yield value of 0.5 t ha^{-1}yr^{-1} in Fafo is caused by its large catchment size.

4.5.6 Comparison between simulated and measured sediment yield

Sediment yield simulated by WaTEM/SEDEM is compared with observed sediment yield derived from stage-volume-curves and sediment core analysis of the reservoirs (section 4.3.5). Results show that simulated sediment yield is in close agreement with observed yield (Figure 4.20). Although the number of three samples is neither representative enough to derive a clear generalized relationship nor sufficient for statistical analysis (e.g., RMSE), the scatter diagram might be still useful to illustrate a first trend for the individual sites.

Whereas the model overpredicts simulated sediment yield by about 22 % (SSY 23%) and 26 % (SSY 25 %) for the catchments Dano and Wahable, respectively, it underestimates sediment yield in Fafo by about 37 % (SSY 40 %).



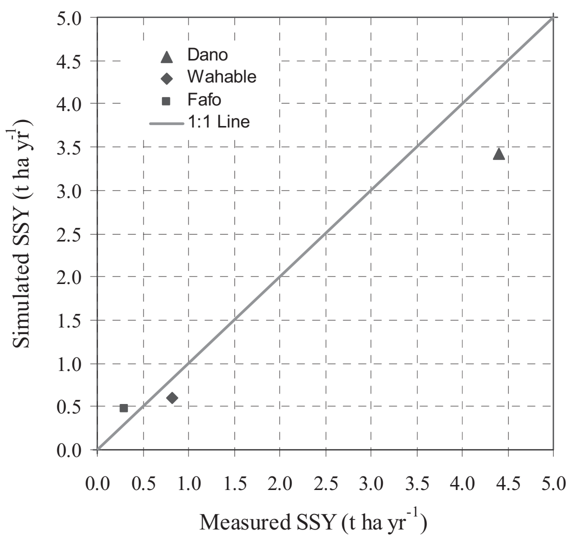


Figure 4.20 Comparison between simulated and measured absolute sediment yield (SY) and area-specific sediment yield (SSY)

These over- and underestimation errors could be due to uncertainties related to the sediment delivery ratio of the model. The sediment delivery ratio in WaTEM/SEDEM is based on a transport capacity equation and a sediment routing algorithm, which assumes that all sediment entering the flow network is transported through the catchment to the outlet of the reservoir. Hence, sediment deposition in channels and flows is not considered. This might lead to an overprediction of simulated sediment yield especially for a comparatively larger catchment with ephemeral flows and low relief energy, such as the Fafo catchment. The lower sediment delivery ratio in the smaller Dano catchment, which is expressed by a higher accumulation rate within the catchment (=sediment deposition rate excluding ponds, section 4.5.5) would support this assumption.

However, differences in simulated and observed sediment yield could also be attributed either to the modeling approach or to the measuring technique. The modeling approach might involve limitations related to both internal shortcomings of the model itself and external uncertainties of parameter estimation. Internal shortcomings of WaTEM/SEDEM might be mainly ascribed to the USLE-based approach, which is one of the most used but also most criticized approaches due to its empirically based factorial equations and its limitations regarding simulation of sediment deposition (Boardman, 2006; Kinnell, 2010; for advantages and limitations of (R)USLE see also section 2.2.3). Nevertheless, WaTEM/SEDEM has incorporated important changes compared to the original (R)USLE, such as the advanced two-dimensional LS-factor that replaces the one-dimensional slope-length factor with a unit contributing area and thus allows accounting for interrill/rill erosion and gully erosion as an effect of flow convergence (Desmet and Govers, 1996). Furthermore, WaTEM/SEDEM overcomes the USLE restriction concerning simulation of only soil loss by integrating a transport capacity equation to predict sediment delivery for each landscape cell (Van Rompaey et al., 2001; Verstraeten et al., 2002).

External uncertainties of parameter estimation can be identified for each of the input/factor maps. Especially the vegetation cover map (C-factor), as one of the most important input parameter maps for simulations, is prone to error propagation due to its spatial and temporal resolution. Spatially, the pixel size of the ASTER image might be still not small enough to reflect the fragmented small-scale pattern of farmers fields; temporally, the static image recording time (here end of the harvesting season) might not be capable of considering seasonal land-use/cover changes of the highly dynamic landscape pattern in Burkina Faso. Further limitations are related to the radiometric and geometric correction procedure and the unsupervised classification of remote sensing images in general and for the individual study sites in particular (section 4.3.7).

In terms of the measuring technique, uncertainties may arise from the bathymetric survey, the retrieval of sediment cores or the calculation of sediment yield by dry bulk density and trap efficiency. As the calculation of sediment yield is based on the difference in sediment thickness at two different points in time, the accuracy of total and specific sediment yield depends on the precision of the initial topographical and actual bathymetric map. Whereas the bathymetric map is limited to the measuring precision of the sonar depth sounder, the topographical map relies, in addition to the exactness of the measurements, on the accuracy of an independent fixed reference point to transform relative height values in absolute elevation values. The retrieval of sediment cores might also involve potential sources of errors in terms of measuring inaccuracies due to the precondition of a precise vertical sampling procedure and the withdrawal of an undisturbed soil sample (section 5.4.3). Finally, sediment yield calculation may lead to inaccuracies by converting the volume of the sediment deposits into sediment mass by using the dry bulk density of the sample and the trap efficiency of the reservoir. Although all possible parameters were measured to minimize the error, additional empirical coefficients were required to calculate the specific weight of the deposit and to determine the trap efficiency of the reservoirs (section 4.3.6).

Although the calculation of both measured and simulated sediment yield might be associated with potential sources of errors, results show that reservoir investigations could be used as a valuable tool to validate sediment yield models, e.g., WaTEM/SEDEM. This is also confirmed by Verstraeten and Poesen (2002), who considered sedimentary deposits in small ponds as an alternative approach to assess sediment yield at different scales. They found that the uncertainties involved in this approach are comparable with errors caused by other measuring techniques such as sediment rating curves or suspended sediment sampling. Furthermore, Van Rompaey et al. (2003, 2005) used sediment deposits in small ponds and reservoirs to quantify sediment yield, and also to calibrate and validate the spatially distributed model WaTEM/SEDEM.

The specific sediment yield from the present study is within the range of results obtained from other investigations in Burkina Faso and Mali (Table 4.10). Most of the studies assess sediment yield based on suspended sediment sampling from discharge measurements. Lamachère (2000), for example, estimated a specific sediment yield of 0.4 t $ha^{-1}yr^{-1}$ for a catchment of 44 km² in the Sudano-Sahelian zone of Burkina Faso. For the same climate zone but in south-eastern Mali, Droux et al. (2003) calculated specific sediment yields ranging from 0.03 t $ha^{-1}yr^{-1}$ to 0.5 t $ha^{-1}yr^{-1}$ for catchments covering between 17 km² and 120 km². Higher specific yield rates of 4 t ha^{-1} yr^{-1} to 8 t ha^{-1} yr^{-1} were estimated based on suspended sediment transport by Karambiri et al. (2005) for a very small sub-catchment of 1.4 ha in the Sahelian zone of northern Burkina Faso. For larger catchments, such as the White Volta catchment with a size of 30000 km² and the catchment of Kompienga with about 6000 km², lower rates of 0.07 t ha^{-1} yr^{-1} (Dumas and Claude, 1977) and 0.7 t ha^{-1} yr^{-1} (H.E.R., 1981) were measured. Mietton (1986) calculated rates of 0.8 t ha^{-1} yr^{-1} for the 102 km² catchment of Boulbi, near Ouagadougou, by the retrieval of sediment samples.

Table 4.10 Measured specific sediment yield (SSY) based on suspended sediment load or reservoir investigations in Burkina Faso and Mali

Case study and authors	**Name of the catchment, province or region**	**Agro-ecological climate zone and annual rainfall**	**Catchment size (km²)**	**Specific sediment yield ($t\ ha^{-1}yr^{-1}$)**
Lamachère (2000)	Gourga (Province of Yatenga), northern region, Burkina Faso	Sudano-Sahelian zone, 460 mm;	44	0.4
Droux et al. (2003)	Dounfing, Djitiko and Belekoni (Bamako region), southwestern Mali	South Sudano zone, 1050-1220 mm	18 103 120	0.03 – 0.4 0.2 0.5
Karambiri et al. (2005)	Katchari (Province of Séno); Sahel region, Burkina Faso	Sahelien zone 512 mm	0.01	4.0 – 8.4
Dumas and Claude (1977)	White Volta, central region, Burkina Faso	Sudano-Sahelian zone 400-800 mm	30200	0.07
H.E.R. (1981)	Kompienga (Province of Gourma), eastern region, Burkina Faso	Sudano-Sahelian zone, 720-905 mm	6000	0.7
Gresillon and Reeb (1981)	Bogandé (Gnagna), Tenado (Sanguié), Boromo (Balé), eastern and central-western region, Burkina Faso	North-Sudano and Sudano-Sahelian zone; 724 mm; 900 mm; 1000 mm	92 38 148	0.6 1.9 3.1
Mietton (1986)	Boulbi (near Ouagadougou), Central region, Burkina Faso	Sudano-Sahelian zone, 896 mm	102	0.8
Schmengler (present study)	Dano, Wahable and Fafo; (Ioba province), southwestern region, Burkina Faso	Sudano-Sahelian zone, 956 mm	8 15 24	4.4 0.8 0.3

These results indicate that the high variability in sediment yield might be mainly related to the effects of scale. The inverse relationship between catchment size and sediment yield was also identified by Verstraeten and Poesen (2001) as the most significant factor explaining about 64 % of the variance in area-specific sediment yield. This relationship can also be confirmed in general for most of the studies in Burkina Faso, but catchment size might not be the sole variable explaining the sediment yield potential of a catchment.

Further controlling variables, such as the agro-climatic zone, rainfall characteristics, vegetation-cover dynamics or land-use changes can have also a significant effect on the magnitude of sediment yield. For example, the study of Gresillon and Reeb (1981) indicates that the amount of rainfall might be more decisive for the quantity of sediment yield than the size of the catchment.

However, although these absolute numbers are useful to compare total annual sediment yield between individual catchments and to estimate the potential siltation rates in reservoirs, they cannot provide information about the internal variability of sediment yield or about the sediment sink and source areas within the catchment. Therefore, the spatial distribution of soil erosion and deposition was simulated by WaTEM/SEDEM for each cell of the catchment.

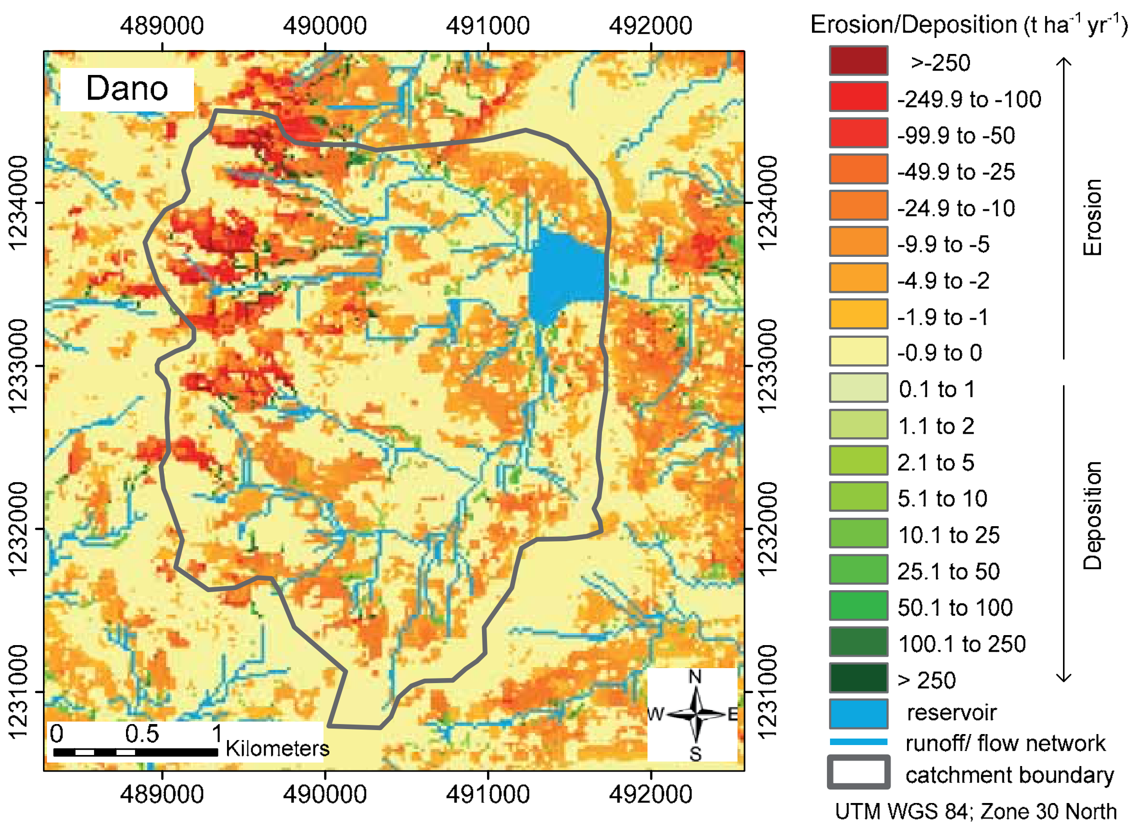
4.5.7 Spatial pattern of soil erosion and soil deposition simulated by WaTEM/SEDEM

The spatial pattern of soil erosion and deposition was simulated by WaTEM/SEDEM firstly to illustrate the magnitude and variability of soil loss and accumulation, and secondly to identify the most severely affected areas, so-called soil erosion hazard zones.

All simulated soil erosion/deposition maps show a high amplitude of simulated values, but expose significant differences between the individual sites (Figure 4.21). While soil loss values range between 1 t ha^{-1} yr^{-1} and 25 t ha^{-1} yr^{-1} in Wahable and Fafo, they exceed 10 times this value in Dano and reach a maximum of 790 t ha^{-1} yr^{-1} on some particularly steep hillslopes of the Ioba Mountains. Deposition values show similar amplitudes, and vary from 1 t ha^{-1} yr^{-1} to 250 t ha^{-1} yr^{-1} in Wahable and Fafo and reach the highest values of more than 250 t ha^{-1} yr^{-1} in some flat areas of Dano.

These extreme values can be explained by the specific terrain characteristics of Dano, where the high slope gradients cause the highest soil loss rates and where the abrupt change between steep slopes and flat areas produce the highest accumulation rates. However, these extremely exposed areas are very small in size, sparsely distributed and cover only a small part of the catchment area. Nevertheless, they require special consideration when soil and water conservation methods are applied in these hazard zones.

The soil erosion/deposition maps show furthermore that most of the catchment area is dominated by very low soil loss rates. At all sites, soil erosion affects approximately 97 % of the total area, and 61 % (Dano), 70 % (Wahable) and 75 % (Fafo) belong to the class of soil loss below 1 t ha^{-1} yr^{-1} (Figure 4.22). Whereas soil loss rates higher than 10 t ha^{-1} yr^{-1} cover less than 1 % of the catchment areas in Wahable and Fafo, they cover more than 13 % of the total area in Dano. However, in contrast to erosion zones, deposition zones have a significantly lower spatial coverage and show a more evenly distributed range in deposition magnitudes.



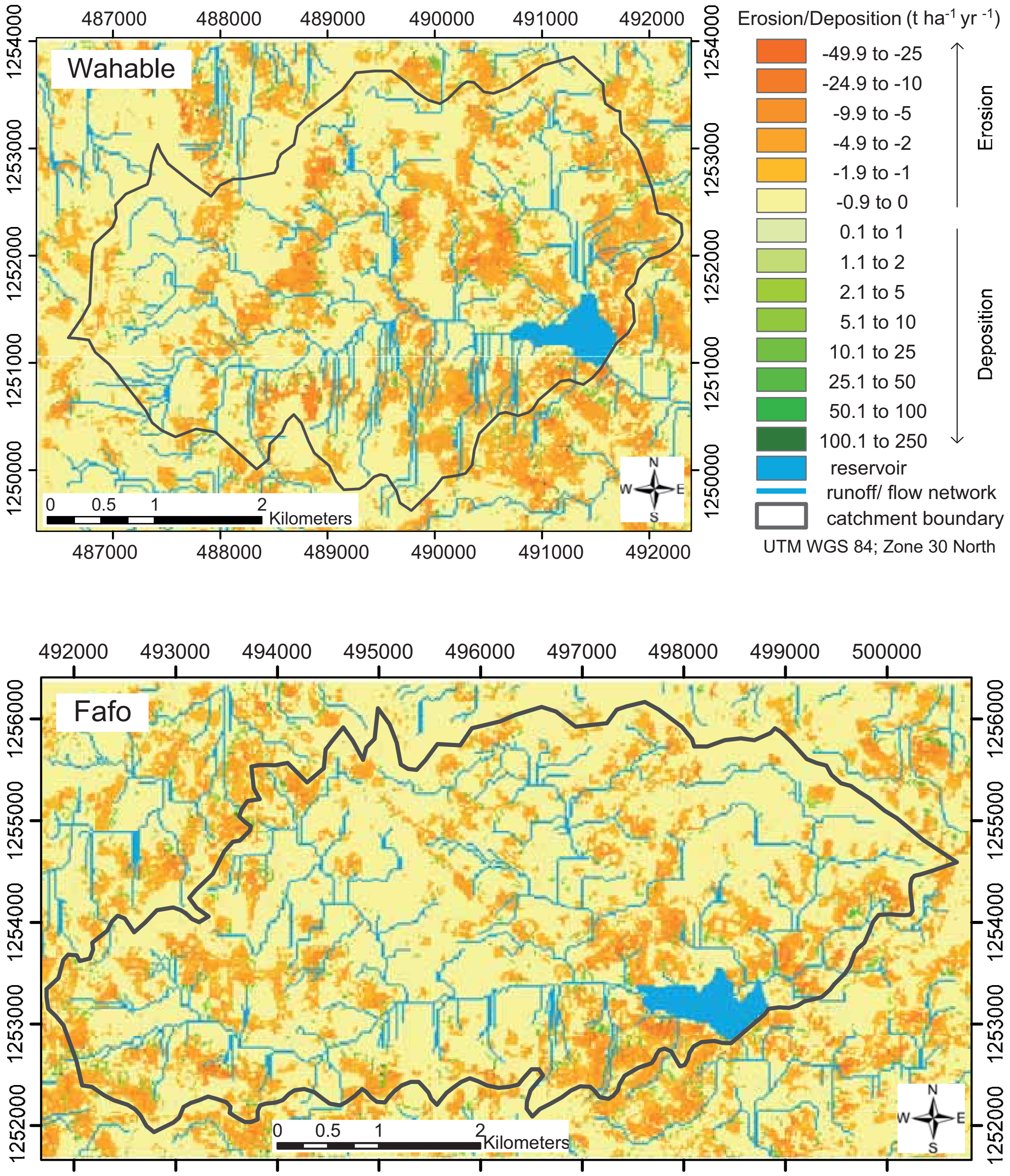
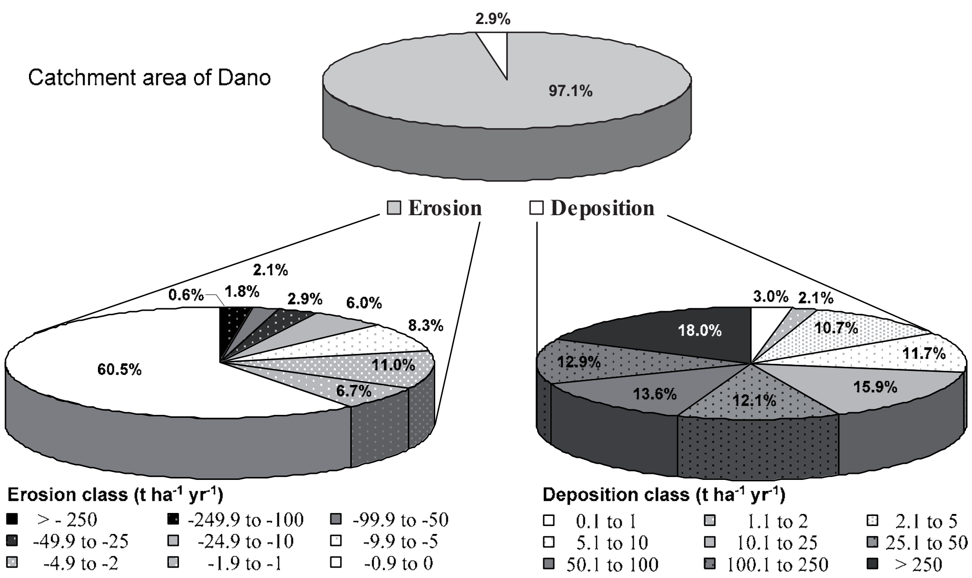


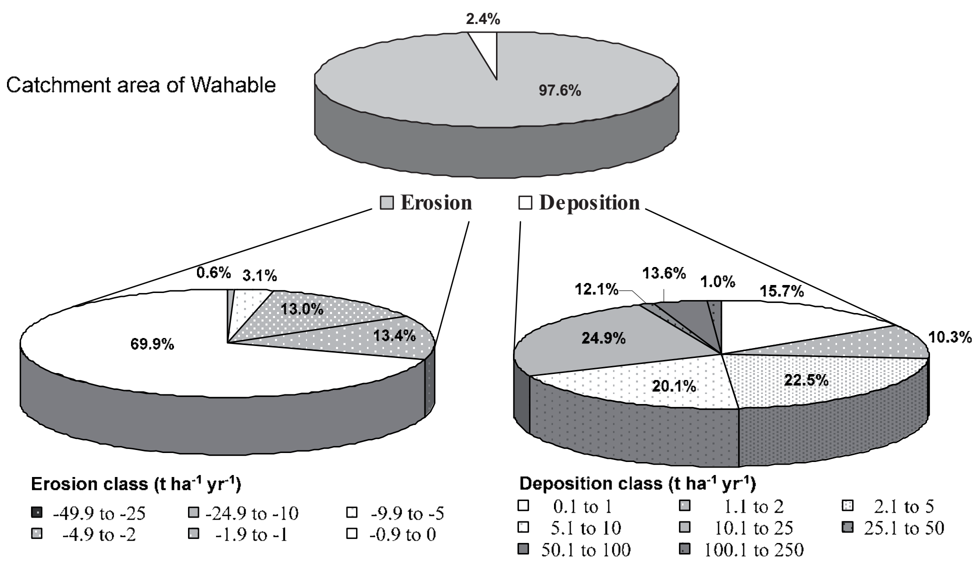
Figure 4.21 Spatial distribution of simulated soil erosion and soil deposition by WaTEM/SEDEM for the catchments of Dano, Wahable and Fafo

In Dano, for example, more than 73 % of the deposition zones experience accumulation rates higher than 10 t ha^{-1} yr^{-1}, and about 45 % of these areas experience values higher than 50 t ha^{-1} yr^{-1}. The relationship is less distinct, but still significant in Wahable and Fafo, where 50 % of the deposition zones gain more than 10 t ha^{-1} yr^{-1} and about 15 %

more than 50 t ha^{-1} yr^{-1}. Low deposition rates of below 1 t ha^{-1} yr^{-1} occur in only 15 % of the deposition areas in Wahable and Fafo and in less than 3 % of the deposition areas in Dano.

Although only a small part of the catchment area is affected, the low spatial coverage but high magnitude of accumulation rates within these deposition zones indicates that, where it occurs, soil deposition is severe. Similarly, simulated values for pond deposition show that a high amount of sediment is accumulated in the reservoirs.





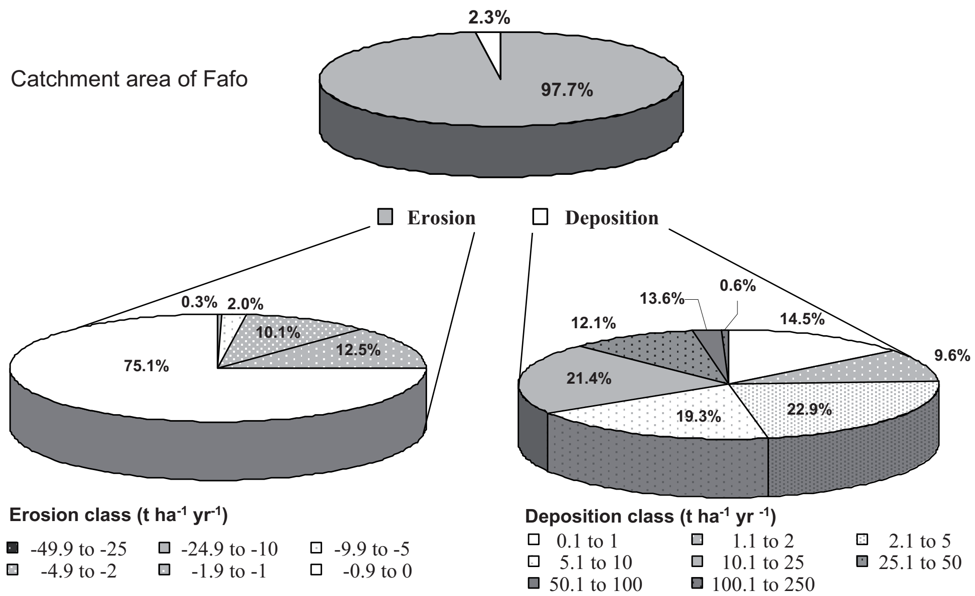


Figure 4.22 Percentage of soil erosion and deposition in relation to catchment area

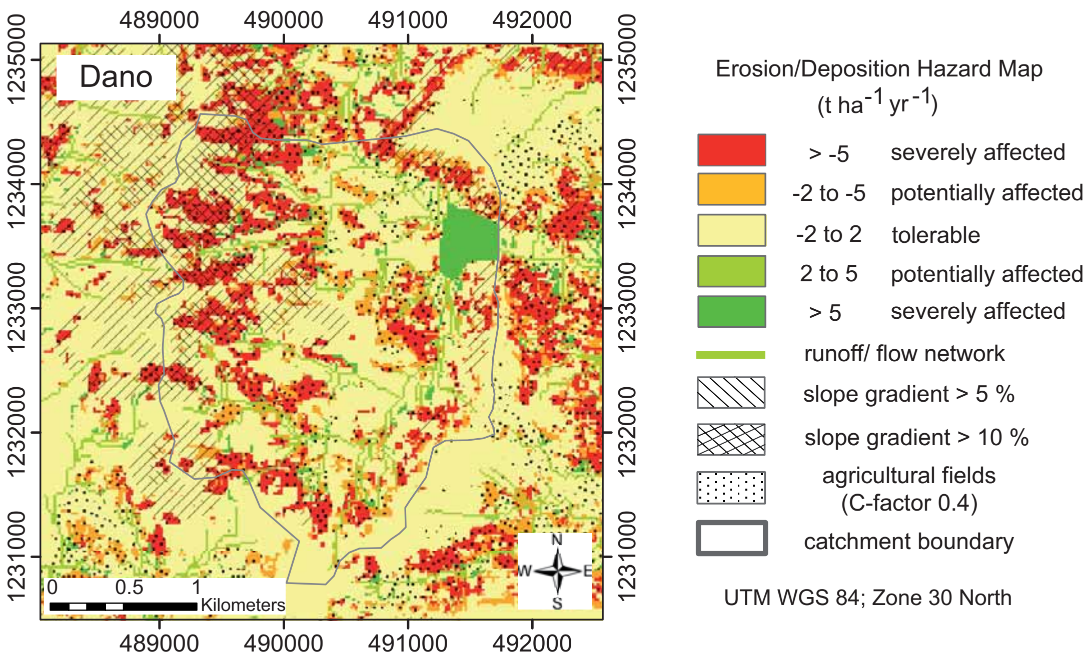
4.5.8 Soil erosion hazard map

Based on the soil erosion and deposition maps simulated by WaTEM/SEDEM, soil erosion hazard maps were generated for each catchment. A threshold value of +/-2 t ha^{-1} yr^{-1} was assigned to potential hazard zones where the tolerable soil loss/accumulation rate was exceeded. The tolerable soil loss rate is defined as the upper limit at which the dynamic equilibrium between soil formation and soil loss is balanced, and the functions of the soil in regard to its productivity and nutrient status are maintained (Li et al., 2009; Roose, 1994; Verheijen et al., 2009). The concept of tolerable soil loss is used as a first valuable benchmark to identify all areas that might be at risk. In these areas, continuous agriculture without additional fertilizer input can lead to land degradation and deterioration of soil quality in the near future.

Additionally, all areas with soil loss rates higher than 5 t ha^{-1} yr^{-1} were identified as severely affected hazard zones. In these zones, soil erosion has led or will lead to considerable soil degradation, reduction in soil productivity and/or deterioration of soil quality on-site and/or off-site. Off-site impacts might include serious soil and water problems. All areas with soil accumulation rates higher than 5 t ha^{-1} yr^{-1} were included as actual soil deposition hazard zones, and all areas with rates higher than 2 t ha^{-1} yr^{-1} as potentially endangered deposition zones.

The hazard map also considers terrain attributes and land management characteristics as determining variables and, moreover, as factors for decision making in terms of soil and water conservation measures. Therefore, all areas with slope gradients higher than 5 %, 10 % and 20 %, and all areas with annual cultivation are illustrated on the map. Annually cultivated fields include all millet or maize fields with C-factors of 0.4. Harvested, burned or abandoned fields, fallow land and irrigated agriculture could not be taken into account due to simplified assumptions for model simulations.

The hazard map of Dano shows that large areas of the catchment are strongly affected by severe soil erosion, and that the tolerable soil loss rate is exceeded in about 30 % of the total catchment area (Fig. 4.23). By including terrain attributes and land management characteristics, the map indicates a clear correlation between erosion-affected areas and intensively cultivated land with slope gradients above 10 %. If, however, only a single criterion, i.e., either intensive cultivation or high slope gradients, is met, the land is not necessarily strongly affected. There are areas in the western part of the catchment and beyond its boundaries that belong to the undulating hillslopes of the Ioba Mountains but do not produce soil erosion rates above the tolerable threshold value. These areas are generally more densely covered by shrubs and sparsely distributed trees.



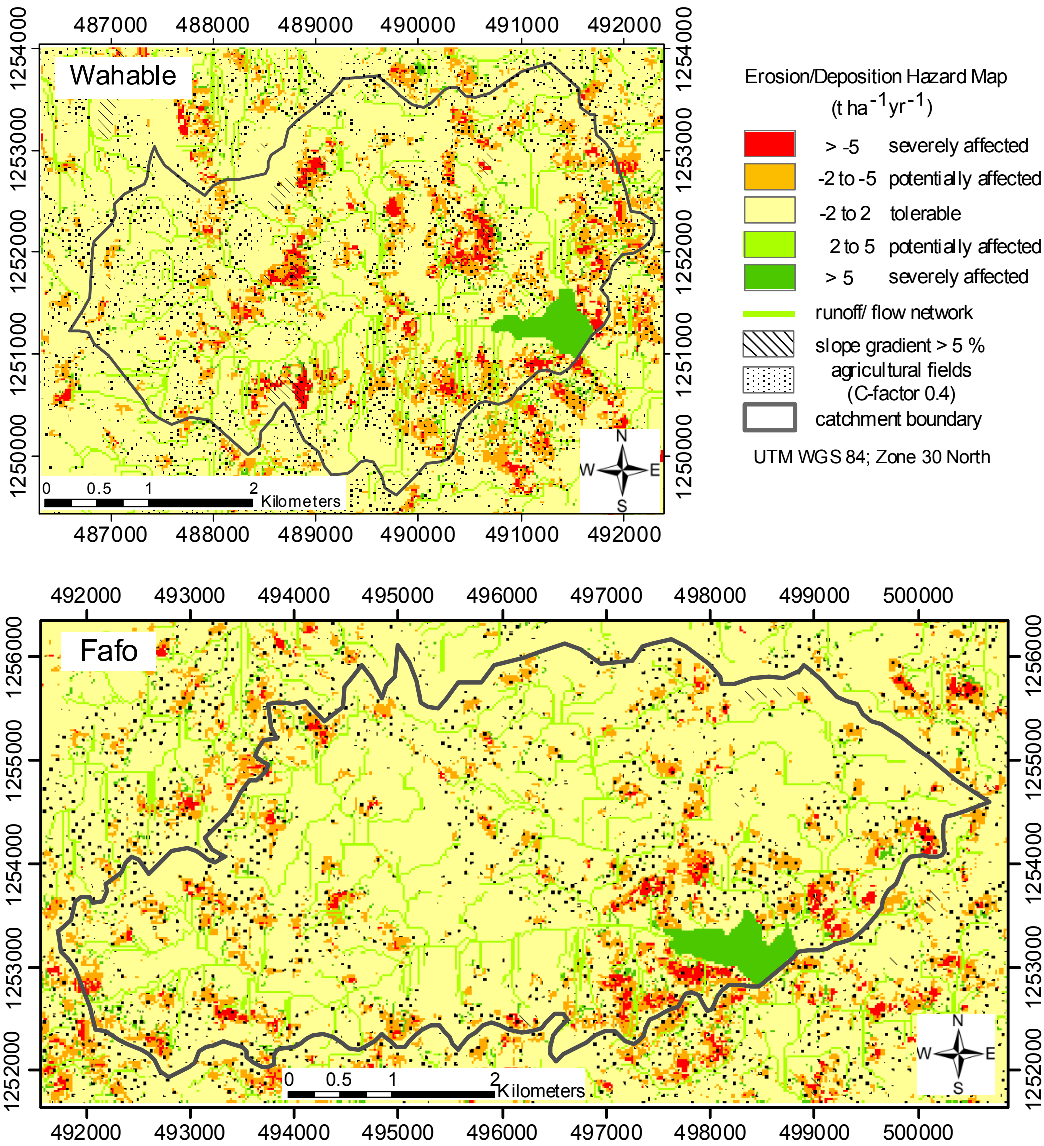


Figure 4.23 Erosion hazard map for the catchments of Dano, Wahable and Fafo

Nevertheless, such areas can become erosion risk zones if the vegetation cover diminishes. On the other hand, intensively cultivated but flat areas often approach the tolerable soil loss rate or are surrounded by potentially endangered zones. This indicates that cultivated fields are highly vulnerable areas, and that vegetation cover and hence land management could have a controlling effect on soil erosion risk.

Deposition hazard zones are mainly limited to the reservoir area, which is highly prone to accumulation, and to the runoff/flow pathways, which are potentially threatened. The runoff network is characterized by perennial flows where transported sediment is accumulated particularly at the end of the rainy season when the magnitude and frequency of rainfalls diminish and the transport capacity is reduced. Also, some single, small deposition patches occur on hillslope areas either close to the runoff network or as a result of the abrupt change in slope gradients.

The catchments of Wahable and Fafo are less threatened by soil erosion and only approximately 16 % and 12 %, respectively, of the catchment areas are affected (Figure 4.23). Due to the near-flat terrain with slope gradients rarely reaching 5 %, the relation between terrain attributes and soil erosion risk zones is not evident. Although almost all highly and potentially affected hazard zones are under intensive agriculture, the reverse is not true. Yet, (potential) erosion hotspots often appear on continuously cultivated fields near rural settlements or along roads/pathways. The villages of Wahable and Fafo are located near the reservoirs, and some homesteads are scattered around the reservoirs. These areas of rural settlement are very prone to erosion, as they are covered by sparse vegetation or degraded bare soil. Continuous cultivation, overgrazing and burning of land near the village frequently leads to pronounced or at least potential erosion hazard zones. In contrast to Wahable and Fafo, the village of Dano is located a few kilometers south of the small catchment, and the few scattered homesteads within the catchment do not yet show such a severe impact of soil erosion. Deposition hazard zones in Wahable and Fafo are mainly limited to the reservoir area and to the runoff/flow pathways. The simulated hazard maps present only a first step in ensuring tolerable soil loss, and should be seen as an initial estimation and not as an ultimate criterion for erosion risk assessment.

The calculation of tolerable soil loss is primarily based on soil formation rates, which can give a valuable threshold value for acceptable soil loss rates (Li et al., 2009; Verheijen et al., 2009). If the equilibrium between soil loss and soil formation is balanced, soil productivity and crop yield are naturally maintained. Two types of soil formations can be distinguished: topsoil formation and weathering of parent material or bedrock. Whereas topsoil formation is a faster process, often assumed to reach about 0.5 cm yr^{-1} (Auerswald et al., 1991), weathering processes are slow and complicated to measure because they occur within the soil profile and vary highly with parent material, soil depth, soil properties and climate. Wakatsuki and Rasyidin (1992) calculated mean weathering rates of 0.7 t ha^{-1} yr^{-1} on a global scale using a geochemical mass balance equation. Additionally, they estimated higher rates of 2.7 t ha^{-1} yr^{-1} for a granitic watershed in southwestern Japan probably as a result of higher annual discharge and acid deposition. Based on a silica mass balance approach, Alexander (1988) determined soil formation rates worldwide between 0.02 t ha^{-1} yr^{-1} and 1.3 t ha^{-1} yr^{-1} (mean=0.5) for non-arable, non-carbonate watersheds with shallow to medium soil depth. Similar global rates between 0.4 t ha^{-1} yr^{-1} and 1.4 t ha^{-1} yr^{-1} (mean=0.7) were documented by Wilkinson and McElroy (2007) in a continental erosion/sedimentation assessment.

Weathering rates for the study site in Burkina Faso were not available, but they are expected to be in the upper range of global soil formation rates due to relatively high chemical weathering rates. Furthermore, variable and high intensive rainfalls as well as wildfires and field-burning practices might have an accelerating effect on weathering rates in the country (Shakesby and Doerr, 2006). Nevertheless, as long as explicit soil formation rates are not known, the precautionary principle should be applied. Therefore, tolerance values for soils in Burkina Faso were assumed to be equal or below mean global soil formation rates and hence below 5 t ha^{-1} yr^{-1} when considering both soil formation rates from topsoil and weathering rates from parent material.

Besides these (in)direct calculations, many additional methods have been developed to determine tolerance values based on soil thickness measurements, soil property analyses or productivity indices (Bhattacharyya et al., 2008; Skidmore, 1982). According to the guidelines of the USDA-Natural Resource Conservation Service (USDA-NRCS, 1999), a tolerance value of +/-2.5 t ha^{-1} yr^{-1} can be assigned to the soils of the study sites in Burkina Faso. This low threshold value is a result of the shallow depth of the root limiting subsurface layer, which is one of the most important criteria for the classification. Field measurements of soil profiles show that the maximum rooting depth is generally less than 50 cm, at some shallow shoulder or backslope positions along the catena even less than 25 cm (section 3.5.1).

Also, following the categories of the USDA-NRCS, the limitations of the soil layer in terms of its physical and chemical properties are significant, and at the same time overcoming these limitations by appropriate land management practices is hardly feasible for economic reasons. Especially the latter is one of the main constrains for subsistence farmers in Burkina Faso, who often neither have the financial means nor the necessary resources to maintain soil fertility or to restore soil nutrients by mineral fertilizers or nutrient recycling. The low economic feasibility and the limited application of fertilizers to compensate nutrient losses was thus the main reason for defining an even lower minimum potential threshold value of +/-2 t ha^{-1} yr^{-1} for the study sites.

The definition of two or more tolerance values for a specific site is becoming an increasingly common and promising strategy to reflect the multi-functionality of a soil in regard to its given natural properties (e.g., topsoil formation and weathering rate), its quality and productivity status (e.g., maintenance of an optimum level of crop productivity long-term) and its socio-economical implications (e.g., feasibility of agro-economical inputs and compensation of erosion damages by social investments) (Skidmore, 1982; Sparovek and De Maria, 2003; Verheijen et al., 2009). Bazzoffi (2009) and Li et al. (2009) suggest including off-site threats of soil erosion by establishing environmental tolerance values, e.g., an agrochemical application tolerance, a cultivation/irrigation tolerance or a hydrogeological risk tolerance. These functional threshold values could be useful criteria to consider the impact of runoff and sediment transfer for the environment downstream and to avoid a deterioration of the water quality in reservoirs and rivers. The latter has a high relevance for Burkina Faso because the number of reservoirs is high and a deterioration of water quality can cause severe health problems for the rural population (Boelee et al., 2009).

Even though these functional tolerance values could not be incorporated in the present study, they should be considered for future research as a tool to develop a scientific guideline for soil and land-use management, ecological reconstruction and other anthropologic activities (Li et al., 2009; Gündra et al., 1995). The definition of multiple functional threshold values could be also an option to overcome former often well-founded criticism about the ambiguity and arbitrariness of a single tolerance value. However, this tolerance threshold approach is still challenging, as it requires a holistic understanding of the multiple effects and interactions in soil erosion research.

In this regard, the soil erosion/deposition hazard map can be seen as an initial step to obtain information about the spatial distribution and severity of soil erosion and sediment delivery within a catchment. Also, the map could be used to identify adequate locations for implementing soil and water conservation measures and to define a preliminary threshold value for environmental hazard zones on-site and off-site.

4.6 Conclusions

On a catchment scale, both on-site and off-site impacts of soil erosion were assessed to measure and simulate the magnitude of topsoil removal in source areas and soil accumulation in sink areas such as dammed reservoirs. Especially the latter, i.e., the siltation of reservoirs, is seen as a serious environmental threat in Burkina Faso because many of the small reservoirs are severely endangered and might lose their function as essential water storages for domestic use, irrigation and stock watering in the near future.

The results of the different approaches applied in this study indicate that bathymetric/topographical analysis, sediment core sampling including stratigraphic and ^{137}Cs measurement, and the modelling approach can be used supplementary to assess reservoir siltation in Burkina Faso. The bathymetric analysis shows that the water storage capacity of the reservoirs is reduced at annual rates between 0.2 % and 0.4 %. Based on their original capacity, the reservoirs have already lost about 10 % to 15 % of their storage volume at normal pool level and might reach their half-life in about 25 years assuming a continuous siltation under current conditions. Stratigraphical changes and down-core variations in sediment properties were compared with estimates from the bathymetric survey. The results show that accumulated sediment layers on the reservoir bed had an average thickness between 0.3 m and 0.5 m, which corresponds to an average annual gain of 1.5 cm and 2.5 cm in the sediment layer. The reference depth profiles of the reservoirs showed significant stratigraphic changes once the in-situ soil of the reservoir bed was reached. Depending on the reservoir characteristics, this abrupt change in soil texture was visible at a soil depth between 25 cm and 40 cm. Measurements of ^{137}Cs confirmed a similar depth distribution by a clear decline in ^{137}Cs concentration from more than 70 % in the accumulated allochthonous soil to less than 30 % in the in-situ autochthonous soil. Results indicate furthermore that variation in sediment thickness might be related to the inverse relationship between catchment size and sediment yield. The thicker sediment layer was found in the reservoir with the smaller catchment size, which might be the result of the smaller contributing area leading to comparatively faster sediment transport and hence faster deposition in the reservoir.

However, given the low water levels of often less than 2 m in the dry season, the relatively high gain in sediment thickness might lead to increased evaporation losses and continuously reduced water use potential in the dry season, to faster and more extensive flooding of the shallow bank areas and to increased siltation in irrigation channels or the dammed outlet.

The simulations of WaTEM/SEDEM show that predicted sediment yield is in close correlation with observed values from the bathymetric survey and the sediment core analysis. Specific sediment yield varies from 0.5 t ha^{-1} yr^{-1} to 3.4 t ha^{-1} yr^{-1} for catchments between 800 ha and 2400 ha. Here, too, the inverse relationship between sediment yield and catchment size can be confirmed. The magnitude of specific sediment yield is within the range of values from other studies in West Africa and can be seen as representative for the semi-arid climate and the environmental conditions in Burkina Faso. In comparison with sediment yield worldwide, the values belong to a medium yield class. Nevertheless, the sedimentation rate is a serious threat considering the importance of small reservoirs for Burkina Faso as reliable long-term water storages in the dry season.

Furthermore, the spatial pattern of soil erosion could be mapped by WaTEM/SEDEM, and the most affected soil loss and soil accumulation zones could be identified. All areas with soil loss or soil accumulation rates exceeding +/-2 t ha^{-1} yr^{-1} are considered as potentially at risk, all areas with soil loss rates higher than +/-5 t ha^{-1} yr^{-1} as already severely affected. The hazard maps show that approximately 15 % to 30 % of the total catchment area is potentially or severely endangered. In these areas, the tolerable soil loss/accumulation rate is exceeded and, moreover, the dynamic equilibrium between soil formation and soil loss is no longer balanced. This implies that soil erosion has led to or will lead to further soil degradation, reduced soil productivity and deterioration of the soil/water environment on-site and off-site if no conservation methods are implemented. The model simulations also show that slope and land-use are often the most decisive factors influencing soil erosion risk, which indicates at the same time that land management could have a controlling effect on soil loss. As potential erosion hotspots often appear on cultivated fields near rural settlements and along pathways, soil and water conservation techniques should be located there. Also, to reduce soil erosion on-site in source areas is more effective and more promising than to deal with siltation problems off-site such as removing sediment from reservoirs (e.g., by hydraulic dredging or dry excavation), which is frequently extremely costly and not feasible in developing countries such as Burkina Faso.

Since the degree of reservoir siltation could be quantified and the extent of soil erosion hazard zones could be localized in this study, further research can build on these results and support the implementation of soil and water conservation techniques in erosion-prone areas in order to minimize or at least to decelerate the siltation behind small dams.

5 CONCLUSIONS AND RECOMMENDATIONS

In this thesis, soil erosion has been confirmed to be a serious environmental hazard in Burkina Faso both due to losses of nutrient-rich topsoil from farmer's fields and due to the rapid siltation of small reservoirs. The assessment of soil loss, soil redistribution and sediment delivery has shown that different pedological, morphological and hydrological methods, remote sensing techniques and soil erosion models are required to estimate on-site and off-site impacts of soil erosion at hillslope and catchment scale.

In accordance with the main research objectives, first the magnitude and spatial patterns of soil erosion and soil accumulation were analyzed at hillslope scale; second, the effects of land management options on soil loss were simulated; third, water storage capacity, siltation rates and sediment yield were calculated at catchment scale and finally, a soil loss and deposition hazard map was simulated and a soil loss tolerance threshold was proposed at the catchment scale.

5.1 Magnitude and spatial variability of soil erosion

Simulated and measured soil loss results indicate that soil erosion is a severe agricultural problem for farmers depending on soil as a resource for their livelihood and lacking often the required means to counteract nutrient loss by soil erosion. The dimension of soil erosion is most adequately assessed by considering its spatial variability within the landscape. Whereas averaged or net soil loss rates can be below or within the range of acceptable values, maximum rates might exceed these values by a multiple. At the hillslope sites in southwestern Burkina Faso, for example, net erosion rates measured by the ^{137}Cs method are below 5 t ha^{-1} yr^{-1} for the entire hillslope, but reach more than 50 t ha^{-1} yr^{-1} at individual hotspots. Simulated results by the Water Erosion Prediction Project (WEPP) model are even more striking and show up to forty times higher soil loss rates at steeper hillslope sections (e.g. backslope positions) compared to averaged soil loss rates when treating the hillslope as a lumped unit. At catchment scale, similar results are obtained by predictions with the raster-based soil erosion and sediment delivery model WaTEM/SEDEM. Simulated soil-loss maps for the catchments of Dano, Wahable and Fafo show that most of the catchment area (>60 %) is dominated by very slight soil loss of below 1 t ha^{-1} yr^{-1}, another large part is characterized by medium soil loss and only a minor part of the areas (1 % to 13 %)

experience soil loss rates above 10 t ha^{-1} yr^{-1}. Though proportionally small, these areas require special attention to avoid or at least decelerate the formation of laterite crusts and hard pans. These erosion-prone areas occur on steeper, remote slope sections but also on continuously cultivated fields near rural settlements, roads, pathways and overgrazed or burned land. In those areas, especially farmers are affected which calls for adjustments in land management and adequate soil and water conservation techniques.

As demonstrated, the development of spatially-explicit hazard maps can serve as a valuable tool to identify present and potential soil-erosion risk zones where the tolerable soil loss/deposition rates are exceeded. Also, by determining maximum and minimum erosion rates, benchmark information is provided to evaluate the magnitude and spatial severity of soil erosion and sedimentation both on individual fields and for larger landscape units.

5.2 Measures to counteract soil erosion by site-specific land management

The selection of appropriate land management methods in Burkina Faso depends not only on the most adequate agronomic, biological or mechanical options but also on the feasibility for farmers to practice this method in terms of implementation costs, labor, time and effort, know-how and their willingness to change long-established cultivation habits.

The traditional and commonly used tillage by handhoe has been proved to be a low cost, low input and hence very feasible farming practice which leads to slight to medium soil loss rates. However, handhoe tillage is very labor intensive and does not improve the soil structure nor ameliorate the nutrient status of the soil. Biologically effective means to ameliorate the soil would be mulching or residue addition in order to replenish the humus layer with soil organic matter and to prevent the soil from being eroded by reducing the splash effect of raindrops. Simulations have shown that a slight straw mulch at the rate of 0.3 kg m^2 could reduce soil loss rates by up to 45 % on erosion-prone hillslope positions. Nevertheless, the use of mulching techniques is limited as farmers can often not provide sufficient quantities of residue. The cost of using artificial mulch, if the awareness is even there, is often prohibitive. Similar constraints apply to the construction of stone-lines as a mechanical obstruction, though they are indeed known to be highly effective in reducing soil loss rates by 85-95 %. Investment funds, transport facilities, labor and know-how are generally in short supply. Detrimental methods of land preparation are the burning of fields and the use of highly mechanized tillage techniques, such as chisel plow, which can provoke up to 300 % higher soil loss rates than those estimated with handhoe tillage. Especially burning is widely used with the aim of clearing of reclaimed or fallow land, natural weed and pest control or the rapid access to stored nutrients locked up in the vegetation. However, burning accelerates not only soil erosion due to a reduced vegetation cover but causes also volatilization losses of soil nutrients and thus diminish soil fertility if annually practiced without adequate fallow periods.

Simulations indicate, that the erosion-risk can be significantly reduced by applying site-specific measures at erosion-prone hillslope sections in particular. The construction of stone lines as a physical-mechanical option to control runoff and the use of residue as a biological-agronomical option to restore soil nutrients would minimize soil loss rates at least by 50 %. A combination of both measures could provide additionally synergetic effects and would meet the requirements of sustainable, site-specific land management.

5.3 Quantification of sediment yield and severity of reservoir siltation

Reservoir siltation presents one of the most important off-site impacts of soil erosion in Burkina Faso where a large number of small dams have been built to store rainfall and runoff water in the rainy season which is dearly needed for domestic use, irrigation and stock watering in the dry season. The accumulation of sediments from upstream in the dammed reservoirs downstream leads to a rapid siltation and hence a continuous decrease in their water storage capacity. In analyzing the stage-storage and stage-area curves of two small reservoirs in southwestern Burkina Faso, a comparison between the initial and actual reservoir bed morphology has shown that both reservoirs have already lost approximately 10 % to 15 % of their original storage capacity at normal pool level and more than 60 % of their initial inactive storage volume at spillway level in the last 15 to 20 years. During this period, inflow sediment had accumulated at the bottom of the reservoir to a thickness of 0.3 m to 0.5 m. Sediment core measurements confirmed this thickness by showing a clear stratigraphical change in soil properties and significant variations in the ^{137}Cs concentrations at these specific depths, which allowed differentiation of the accumulated, allochthonous sediment from the in-situ, autochthonous soil. The calculation of sediment yield was used as an additional indicator to quantify sediment delivery and sediment deposition at the outlet of the catchment. Specific sediment yield ranged from 0.5 t ha^{-1} yr^{-1} to 3.4 t ha^{-1} yr^{-1} and was higher for the smaller catchment.

Although these values can be seen as relatively low in comparison with sediment yield studies worldwide, they are serious for the semi-arid environment of Burkina Faso where the rural population is dependent on small reservoirs as reliable water storage in the dry season. Furthermore, the severity of siltation becomes even more obvious if considering that a time span of about 25 years is expected before the “half-life” of both reservoirs is reached and siltation might seriously affect its water supply and flood control.

5.4 Methodological background and multi-scale approaches: uncertainties, potential and limitations

Different scale-dependent methods and models were used to simulate soil erosion and sediment deposition at hillslope and catchment scale. The hierarchical approach was chosen to account for the dominant environmental processes at each of the spatial scales individually and to avoid uncertainties related to upscaling and downscaling of parameters.

At hillslope scale, the WEPP model served as a physically-based erosion model to predict average annual soil loss rates along slope profiles based on the current environmental conditions. The soil loss rates derived from the remaining and still detectable concentration of the radionuclide ^{137}Cs in the soil was found to be comparable with these simulated results. Although model predictions and ^{137}Cs estimates indicate that soil loss is high and will lead to a removal in the near future of the A-horizon at erosion-prone hillslope sections, the agreement between both methods is low. The low correspondence in absolute soil loss values could be a result of the shortcomings of each approach, including internal model limitations, measurement errors or uncertainties associated with the spatial and temporal variability of input parameters. Besides general constraints of physically-based models, the most significant limitations of the WEPP model in this study were the high input data requirement especially in terms of climate and land-use parameters, the overestimation of soil loss for clayey soil and the negligence of vertical pedogenesis. In particular the latter might be relevant for soils in Burkina Faso where the development of illuvation horizons, plinthite layers or soft bedrock is common and its influence on the soil-water balance should be reflected in the model. Limitations due to the lumped, spatially non-explicit character of the model could be overcome by integrating the catenary concept in the modeling approach.

Whereas the WEPP model estimates actual erosion rates, the ^{137}Cs-approach, in contrast, is a retrospective, indirect evaluation of soil loss during the last 50 years. Thus the two results are not necessarily expected to correspond. The accuracy of the ^{137}Cs measurements is highly dependent on the selection of suitable reference samples, which might be difficult to locate. Detailed information about former soil conditions and historical land-use, which relies often solely on farmers memory, is often lacking. Also, some limitations might be associated with the assumption of static parameter values for the conversion models, such as the relaxation depth and the particle size correction factor. Furthermore, both models consider water to be the only driving agent of soil erosion and do not account for erosion by wind. Nevertheless, the WEPP model as well as the ^{137}Cs approach have been proven to be applicable in the environment of Burkina Faso and were found capable to provide valuable estimates of current and historic soil loss and soil deposition, respectively, for individual hillslopes.

At the catchment scale, bathymetric surveys, sediment core analysis and ^{137}Cs measurements were used to quantify the magnitude and thickness of accumulated sediment at the reservoir bed. In comparing these observed results with sediment yield values predicted by the WaTEM/SEDEM model, a high agreement was found. However, even if a close correspondence can confirm the potential of these methods to serve for calibration and/or validation purposes, uncertainties and possible sources of errors should be known in advance and considered with care. The bathymetric analysis, for example, depends on the accuracy of the initial topographical and actual bathymetric map and is therefore limited to the measurement precision of both maps and to the availability of a common reference point to enable an accurate comparison between changes in reservoirs morphology. The method, in general, presents a practical approach to reservoir investigations in Burkina Faso, as precise topographical construction maps are available for many dams and actual morphological maps can be easily derived from water depth measurements by a sonar depth-sounder or by a simple stadia rod.

The retrieval of reservoir bed samples by an under-water sediment core sampler has been proved to be a suitable tool to validate the results of the bathymetric survey. This approach, however, is more complicated and might involve uncertainties related to the withdrawal of undisturbed sediment samples, the interpretation of stratigraphical changes and the conversion of sediment volume into sediment mass. The sediment cores can be additionally used for the analysis of ^{137}Cs concentration with depth. A sufficient amount of ^{137}Cs is still detectable in reservoir soils even if the dams have been built long after the peak of the radionuclide fallout in the 1950s.

Simulations by the soil erosion and sediment delivery model WaTEM/SEDEM have shown that the empirically-based, but spatially explicit structure of the model was well suited to predict soil erosion and sediment yield in Burkina Faso. Nevertheless, considerable limitations are associated with internal shortcoming of the model and its USLE-based approach. Further uncertainties may arise from the parameter estimation for the input factor maps, especially for the digital elevation model map and the land-cover map when based on an (un)supervised classification of remote sensing images. Also, sediment yield predictions for the catchments in southwestern Burkina Faso indicate that the model overestimates sediment yield for the larger and flatter catchments, which might be due to limitations related to the sediment delivery ratio.

The use of multiple methods at different scales allows evaluation of the potential and limitations of the individual methods more closely, facilitates calibration and validation procedures and thus enables a more accurate estimation of soil erosion and reservoir siltation especially for ungauged catchments.

5.5 Recommendations and future research needs

Based on the assessment of soil erosion at hillslope and catchment scale in Burkina Faso, the following recommendations and future research needs are suggested:

- **Consideration of scaling issues in environmental modelling:**
  Environmental processes should be considered within the specific spatial and temporal dimension in which they operate, in which they can be measured and finally in which they are captured for modeling. The consideration of these phenomenological, methodological and modeling scales is important for finding new pathways to transfer knowledge between systems, to identify scale dependencies of parameters, to define scaling relations of processes and to reflect feedback mechanisms between different scales. Hierarchical or nested approaches present a way forward to account for the most dominant processes at each scale, but further research is needed to discover synergy effects between components, to simulate nonlinearities by models and to account for the emergent properties within complex environmental systems.
- **Catenary concept as a framework for erosion modelling:**
  The systematic sequence of chemical and physical soil properties along slope profiles confirmed that the catena concept is both observable for soils in Burkina Faso and a useful gain for soil erosion modelling. If soil loss models integrate the catenary soil sequence in their modeling structure, e.g., by dividing the hillslope into distinct soil-landscape units, they can better consider the spatial variability of soils and can more clearly demarcate soil erosion and sedimentation zones. The integration of the catenary concept presents furthermore a promising spatially-explicit framework to adjust soil and water conservation methods to the specific requirements of individual hillslope sections.
- **Integration of nutrient components in soil erosion models:**
  Integrated nutrient management is an essential requirement to enhance crop yield and to ensure food security especially in environments where soils are highly deficient in primary nutrients. Future soil erosion models should therefore couple, link or integrate plant nutrient models with soil erosion models in order to quantify nutrient loss as a consequence of soil erosion. Furthermore, to select more adequate land management techniques to prevent and restore soil nutrients, simulation

scenarios should reflect soil conservation and nutrient management options, which are also feasible for farmers in West Africa (e.g., construction of stone rows, mulching, residue addition, fertilization).

- **Need to evaluate off-site impacts and to assess sedimentation of reservoirs:** There exists a great necessity to analyze the off-site effects of soil erosion, such as the siltation behind dams, which have been rarely documented for the large number of small dams in Burkina Faso. Further research should therefore quantify the magnitude of siltation, monitor its influence on losses in water storage capacity and estimate the life expectancies of dams. An accurate assessment of reservoir sedimentation should be made already during the planning of new dams to avoid severe siltation problems. The importance of such studies is especially given in semi-arid countries such as Burkina Faso, where farmers rely on sustainable water resources for their livelihoods.
- **Establishment of a soil erosion hazard map for decision makers:**
  Soil erosion hazard maps present a powerful tool to define soil loss tolerance values for environmental hazard zones on-site and off-site and to identify the best locations to implement soil and water conservation measures. Additionally, an erosion hazard map at regional or national scale could serve to support policy makers in their decisions on which landscapes are primarily at risk and where action plans should be implemented first.

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