Multi-scale interactions between soil, vegetation and erosion in the context of agricultural land abandonment in a semi-arid environment

University of Amsterdam, IBED
Amsterdam, The Netherlands

Cover design: Jan Peter Lesschen

Printed by Wöhrmann Print Service, Zuthpen, The Netherlands

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ACADEMISCH PROEFSCHRIFT

ter verkrijging van de graad van doctor
aan de Universiteit van Amsterdam
op gezag van de Rector Magnificus
prof. dr. D.C. van den Boom
ten overstaan van een door het college van promoties
ingestelde commissie,
in het openbaar te verdedigen in de Agnietenkapel
op vrijdag 26 september 2008, te 12:00 uur

door

Jannes Peter Lesschen

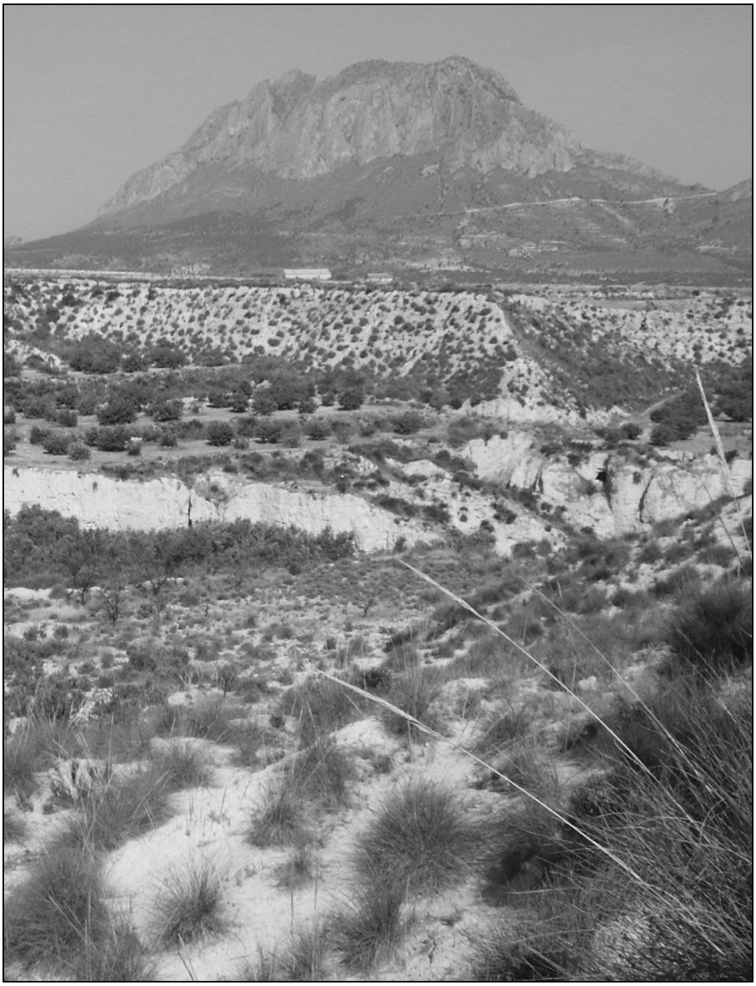
Geboren te Oosterhesselen

Promotor:	prof. dr. J.M. Verstraten
Copromotor:	dr. L.H. Cammeraat
Overige leden:	prof. dr. ir. W. Bouten prof. dr. J. Sevink prof. dr. J. Hooke prof. dr. B. van Wesemael prof. dr. ir. A. Veldkamp prof. dr. S.M. de Jong dr. A.C. Seijmonsbergen

Faculteit Natuurwetenschappen, Wiskunde en Informatica

This research was carried out at the research school Centre for Geo-ecological Research (ICG) and the Institute for Biodiversity and Ecosystem Dynamics (IBED), Earth Surface Processes and Materials, University of Amsterdam, The Netherlands. This research, within the RECONDES project, was funded by the European Commission, project no. GOCE-CT-2003-505361.

Contents



1. Introduction

1.1. Research framework

The title of this thesis is "Multi-scale interactions between soil, vegetation and erosion in the context of agricultural land abandonment in a semi-arid environment". This title represents the three main themes of this thesis, which are soil erosion, scale issues and agricultural land abandonment. Soil erosion is one of the main environmental problems in Mediterranean countries which increases desertification and results in soil quality loss and off-site effects such as flash floods and reservoir sedimentation (Poesen and Hooke, 1997). For the mitigation of soil erosion it is important to understand the mechanisms and the critical soil conditions that are necessary for maintaining and restoring soil quality. Vegetation is one of the key factors that controls soil erosion and which can be used for the mitigation of soil erosion by reducing the connectivity of water and sediment (Thornes, 1990). The second theme is scale, which has a spatial as well as temporal dimension and together they determine which erosion processes are relevant. The spatial scales in this research range from plot to a third order catchment and the temporal scales from minutes to decades. One of the aims of this thesis is to link plot scale observations with catchment scale erosion modelling. The third central theme of this thesis is agricultural land abandonment, which is one of the main changes in land use in marginal areas of northern Mediterranean countries. However, about the consequences of land abandonment not much is known. On the one hand an increase in vegetation cover can decrease erosion, but on the other hand existing soil and water conservation structures are no longer maintained, which can increase erosion. Semi-natural landscapes in dynamic equilibrium are considered to be stable and will have relatively low erosion rates. However, changes in land use such as agricultural land abandonment will disrupt the (in this case artificial) equilibrium. Physical and biological processes will adapt to the new situation, which can lead to an increase or activation of erosion processes, e.g. the spontaneous reorganisation of no longer preserved artificial drainage networks (Gallart et al., 1994). These three themes will be further discussed in the background section of in this introduction and will come back in the different chapters of this thesis.

This research was carried out at the University of Amsterdam and was part of the RECONDES project, a three year EU-project coordinated by the University of Portsmouth. Other partners in the project were the Katholieke Universiteit Leuven and the Université Catholique de Louvain from Belgium, CSIC-CEBAS from Spain and CNR-IRPI from Italy. The focus of RECONDES was to address the mitigation of desertification processes by the means of innovative techniques using vegetation in specific landscape configurations prone to severe degradation processes. Its major objective was to produce practical guidelines on the conditions for use of vegetation in areas vulnerable to desertification, taking into account spatial variability in geomorphological and human-driven processes related to degradation and desertification. Most of the research was carried out at the Carcavo basin, a 30 km^2 catchment in Southeast Spain.

Soil erosion is a very broad research field with inputs from many different disciplines, ranging from meteorology for the analysis of extreme rainfall events to social sciences for the study of socio-economic factors that determine land use dynamics. Four main factors can be distinguished that influence erosion: erosivity (rainfall characteristics), erodibility (soil properties), slope (topography) and plant cover (Morgan, 1995). However, these factors are determined by many different physical, biological and socio-economical processes that occur on different spatial and temporal scales. Since one cannot study all aspects related to soil erosion a clear definition of the research framework is essential. Primarily the hierarchical framework of the RECONDES project is followed with a focus on the erosion processes that occur on abandoned land. However, the other land units were included for the catchment scale component of this thesis, but processes within these land units were not studied in detail. Soil erosion processes on agricultural land were studied in more detail by the University of Louvain (e.g. Van Wesemael et al., 2006), erosion in channels was studied by the University of Portsmouth (e.g. Sandercock et al., 2007) and the effects of plant roots on rill and gully erosion was studied by the University of Leuven (e.g. De Baets et al., 2007). Hence, the main focus of this thesis will be on soil erosion processes on abandoned land at multiple scales. The concept of hydrological connectivity (Bracken and Croke, 2007) will be used to link the different spatial scale levels that range from plot to catchment.

1.2. Background

1.2.1. Soil erosion

Spain is one of the countries that is most severely affected by soil erosion in Europe. The presence of highly erodible soils, a relatively steep topography, periods of drought and torrential rainfall explain the high erosion risk in Spain (Solé Benet, 2006). Additionally, a long history of anthropogenic disturbances such as deforestation and agriculture on marginal lands accelerated soil erosion processes and led to the loss of well-developed soils in the Mediterranean. Nowadays many mountain slopes are bare and valleys are filled with sediment (Yaalon, 1997). In the Region of Murcia 46 percent of the total area was classified as soils with high or very high erosion risk (Región de Murcia, 2002). Soils in the semi-arid parts of the Mediterranean are especially vulnerable to erosion because of low vegetation cover and the occurrence of high intensity rainfall events. Kosmas et al. (1997) showed that runoff and sediment losses on hilly Mediterranean shrublands increased with decreasing annual rainfall, until a maximum at 280-300 mm, which could be attributed to a decrease in vegetation cover. In semi-arid areas overland flow normally occurs when rainfall intensity exceeds the rate at which water infiltrates into the soil (Hortonian overland flow). Once flow becomes concentrated its velocity and erosive force increase and rill and gully erosion become the main erosion processes. Recent studies indicate that gully erosion is one of the main erosion processes in terms of sediment production (Poesen and Valentin, 2003).

The problems caused by soil erosion can be divided in on-site and off-site effects. Soil fertility loss due to the removal of fertile topsoil is the main on-site effect, but also deterioration of soil physical properties, which reduces infiltration and enhances further erosion, is a negative on-site effect. The increased amounts of water and sediment that are transported downwards through the catchment can lead to flash floods and reservoir sedimentation downstream. Flood events in the Mediterranean region are characterized by an extremely rapid rise in the discharge, which is one of the reasons that these events tend to be so devastating and dangerous. These floods are often very localized and typically peak flows decrease downstream from the zone of the storm due to transmission losses (Poesen and Hooke, 1997). For Spain an increase in the frequency of these torrential rainfall events is observed (Alpert et al., 2002). Apart from rainfall characteristics also the physical characteristics of a basin such as steep slopes, sparse vegetation, thin soils and permeable rocks determine the generation of flash floods (Camarasa Belmonte and Segura Beltran, 2001). A second off-site effect is sedimentation of reservoirs and corresponding loss in water storage capacity. The annual loss in storage capacity of many reservoirs due to sediment deposition can be up to five percent, which means that the majority of the capacity is lost after only 25 years. These high rates of storage loss pose a serious threat to the economic sustainability of the reservoir (Verstraeten et al., 2003). For example the Puentes dam in the Guadalentín, about 50 km south of the Carcavo basin, has been raised already three times in the last 150 years due to sedimentation of the reservoir.

In Mediterranean environments the interactions between soil and vegetation have a major influence on soil erosion. Vegetation protects the soil because the canopy and litter intercept raindrops and reduce their kinetic energy. Furthermore, vegetation has a positive influence on soil quality due to the organic matter input by litter. Vegetation also increases chemical weathering, enhances infiltration and favours a less contrasted microclimate. These conditions generate a more active fauna and flora, and consequently soil structure and other soil physical properties such as soil aggregation, water storage capacity and porosity are improved. All these positive influences of vegetation lower the erodibility and decrease the risk of soil erosion (Bochet et al., 1999; Cammeraat and Imeson, 1999).

Vegetation in semi-arid environments is characterised by heterogeneous patterns of bare soil and vegetation patches (Valentin et al., 1999). This mosaic of vegetated and bare zones makes overland flow highly discontinuous as a result of the non-uniform infiltration (Cerdà, 1998; Puigdefabregas et al., 1999). The bare patches between plants function as runoff generating areas, generally bare rock and crusted areas that are characterised by poor soil structure with low infiltration rates, whereas the vegetated patches function as runoff sinks.

To mitigate soil erosion several soil and water conservation practices are applied in the Mediterranean. On agricultural land cover crops are often used to protect the soil against erosion, but also for fertilisation as green manure. Winter wheat and other cereals are most popular. However, under perennials the use of cover crops is very low, because farmers prefer fresh bare soil for higher infiltration and to avoid competition for water (Pastor, 2004). Conservation tillage is therefore hardly adopted in these areas. Moreover, zero-tillage on soils that are vulnerable to crusting, e.g. marly soils, might even enhance erosion due to increased runoff. In stony soils conventional tillage can increase the rock fragment cover at the soil surface, which functions as a mulch layer (Oostwoud Wijdenes et al., 1997). Revegetation is one of the main techniques to control erosion on gullied areas, landslides, road embankments and quarries. To mitigate gully erosion, natural vegetation with well-developed roots should be (re)established in concentrated flow zones affected by gully erosion. This will decrease soil loss and sediment production and the connectivity in the landscape will be interrupted resulting in a lower sediment delivery to valley bottoms or river channels (Poesen et al., 2003). The use of vegetation for mitigation of erosion was also the focus of the RECONDES project. In the Mediterranean terracing is the most widely used measure for soil conservation. The original aim of terracing is to reduce soil erosion and to intercept runoff by decreasing the general slope (Morgan, 1995). However, expansion and mechanisation of tree crop plantations since the nineties lead to the removal of terraces and increased tillage erosion (Van Wesemael et al., 2006; Borselli et al., 2006). In Spain the construction of checkdams in ephemeral channels of low order catchments in combination with reforestation of degraded hillslopes is the traditional way to mitigate erosion by the Spanish Forest Administration. The small dams are supposed to stabilise the channels and reduce erosion. However, besides their positive upstream effects of sediment storage and channel stabilisation, the dams also cause erosion downstream (Castillo et al., 2007).

1.2.2. Scale issues in soil erosion research

In soil erosion research the issue of scale is very important since different processes control erosion at the various spatial as well as temporal scales, which leads to different runoff and erosion rates. Results obtained at one scale are therefore not representative for other scales, since the different processes are not taken into account. For example plot scale studies do not consider gully erosion, while at catchment scale most of the sediment yield is often produced by gully erosion. Erosion processes are therefore generally non-linear and scale dependent. This scale dependency has also lead to a separation in soil erosion research, which can roughly be divided in detailed plot scale studies and general catchment scale studies. Plot scale studies are based on techniques such as Gerlach troughs at open plots, rainfall simulation experiments, erosion pins, USLE based bounded plots, Cesium-137 measurements or laboratory experiments. Whereas catchment scale studies are often based on sediment yields from reservoirs, discharge measurements or remote sensing imagery (Solé Benet, 2006). However, these two types of soil erosion studies are often not combined, which makes that general catchment studies lack the information from relevant process at finer scales, while plot scale studies are often too detailed to be applied at broader scales. The same holds true for erosion models, where detailed erosion models, e.g. EUROSEM or WEPP, require so much input data, that it becomes almost impossible to use these models at broader spatial scales. An overview of European soil erosion models is given by Jetten and Favis-Mortlock (2006), who discussed the model's approaches and the spatial and temporal scales at which they are applied.

Several studies have experimentally demonstrated scale dependency with in general decreasing area specific runoff and erosion rates (e.g. Cammeraat, 2002; Wilcox et al., 2003; De Vente and Poesen 2005). This decrease can be partly attributed to the influences of sinks, i.e. areas of infiltration and sedimentation, such as increased infiltration under vegetation, sediment deposition and infiltration on agricultural terraces and sedimentation in reservoirs. The connectivity between the different sources and sinks of runoff and sediment determines the impact and magnitude of runoff and erosion at broader scales. Also the temporal dimension is important to consider, since extreme events, which have a long recurrence time, are known to contribute substantially to total erosion and landscape formation, particularly in semi-arid landscapes (Boardman, 2006). Only continued monitoring over long time scales can include the effects of these extreme events (Boardman, 2003). Short-term monitoring studies are therefore not very suitable for the assessment of long-term erosion rates.

Schulze (2000) identified six causes of scale problems from a hydrological perspective: spatial heterogeneity in surface process, non-linearity in response, processes require threshold scales to occur, dominant processes change with scale, development of emerging properties and disturbance regimes. These six causes are shortly explained below in the context of soil erosion. Landscapes are characterised by spatial heterogeneity which influences soil erosion processes, this variability is manifested in topography (e.g. slope and position), soils (e.g. infiltration capacity and crusting), rainfall (e.g. intensity and frequency) and land use (e.g. vegetation cover and root type). Erosion processes act at different time scales (minutes to millennia) and with different rates (e.g. splash erosion versus piping), which makes the responses highly non-linear. Runoff generation and erosion processes are subject to thresholds, which are determined by the physical characteristics of a landscape and human interventions (e.g. check dam construction) and these thresholds are scale dependent (Cammeraat, 2004). Furthermore, the dominant erosion processes change with spatial scale, e.g. gully and bank erosion are dominant at catchment scale, while splash and sheet erosion are the main processes at plot scale. Emerging properties can arise from the interaction of small-scale properties, which have a different influence at large scale compared to the small scale, e.g. field boundaries can form important thresholds for erosion at hillslope scale, but the influence at plot scale is insignificant. Disturbance regimes are the last cause of scale problems, for example changes in land use or the construction of dams. All these reasons make upscaling of soil erosion processes very complicated.

Several approaches for scaling of geomorphological processes can be distinguished. However, all approaches have their disadvantages and biophysical processes remain 'pseudo represented', which makes the problem of upscaling still a largely unsolved one (Schulze, 2000). The most basic approach is extrapolation, which is used to scale up point measurements to larger areas under the assumption that spatial variability can be neglected. However, this assumption is generally not valid, especially not for semi-arid areas with their heterogeneous vegetation patterns and non-linear system responses. A second approach is a lumped model which considers the area of interest (e.g. catchment) as a single entity for which the spatial variables (e.g. soils, land use) are averaged and the model parameters are calibrated until the observations are reproduced. However, such a lumped model is only representative for the calibrated conditions and cannot be directly applied to other areas or for other time scales. A third approach is a distributed model in which processes observed at point scale are represented by relatively homogeneous units, called response units or hydrological similar surfaces (England and Stephenson, 1970; Cammeraat, 2002; Kirkby et al., 2002). A disadvantage of this approach is the ambiguity to which degree these response units have to be disaggregated to commence the upscaling. A last approach is modelling at fine enough resolution, however, high data requirements and changing erosion processes with scale make also this approach not the solution for upscaling (Schulze, 2000).

In ecology the hierarchy theory is often used for scaling issues (O'Neill et al., 1986). In this approach a specific scale of interest is selected and finer-scale processes are incorporated in a nested hierarchy. This concept was also used by Cammeraat (2002) in combination with response units. He studied and quantified the hydrological and erosional response of watersheds and its sub-systems based on a nested measurement approach in a first order drainage basin in Southeast Spain. The outcomes demonstrate the strong influence of both spatial and temporal scale on the generation of runoff (Figure 1.1). At plot scale the average threshold for runoff generation was 15 mm of rainfall, while the threshold for runoff generation at watershed scale was about 33 mm. Consequently, the threshold for runoff generation increases with catchment size. Figure 1.1 also shows the temporal dimension of runoff occurrence, which is strongly related to the recurrence period (Cammeraat, 2004). This example of runoff generation demonstrates the importance of both spatial and temporal scale in soil erosion research.

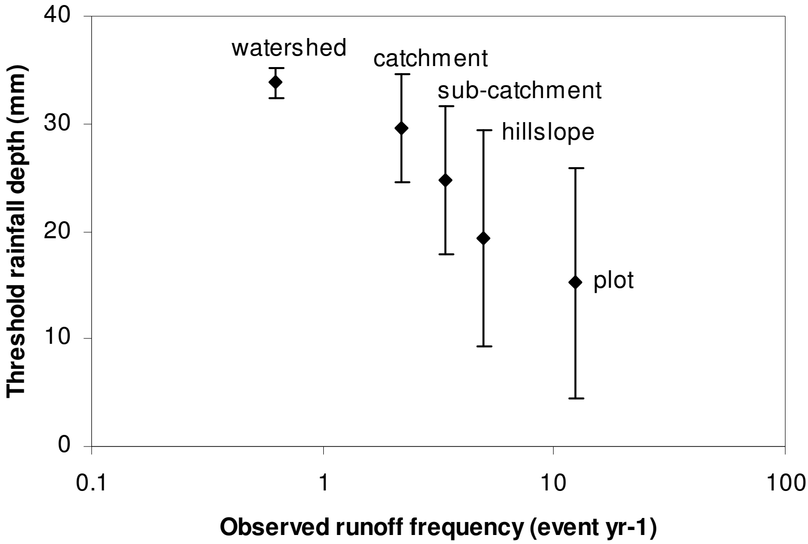


Figure 1.1. Threshold rainfall depth required to generate runoff at different scales within a catchment in Southeast Spain for a three-year period (adapted from Cammeraat (2002) with permission from © John Wiley & Sons Limited)

1.2.3. Agricultural land abandonment

Land abandonment is widespread in Europe and the influence of environmental changes is unpredictable due to environmental, agricultural and socio-economic contextual factors (MacDonald et al., 2000). Although land abandonment occurs in the whole Mediterranean Basin, e.g. France (Taillefumier and Piégay, 2003), Portugal (Pinto-Correia and Mascarenhas, 1999), Italy (Blasi et al., 2000), Greece (Kosmas et al., 2000) and Israel (Neeman and Izhaki, 1996), most literature is available for Spain, where abandonment of agricultural land is nowadays widely spread (Fernandez-Ales et al., 1992; MacDonald et al., 2000; Geeson et al., 2002). Abandoned land is in this thesis defined as areas previously cultivated but now abandoned and where the natural vegetation has been allowed to grow under various intensities of grazing (Kosmas et al., 2002). This definition is used since in most parts of the Mediterranean almost all natural vegetation is grazed to some extent by migrating or permanent flocks of goats and sheep (Clark, 1996).

Already since Roman times land use in Spain has been changing due to human influences. Forest clearing, reclamation and terracing of vast areas around the population centres took place mainly in the 17th and 18th centuries as a consequence of high population pressure (Ruecker et al., 1998). The greatest land use changes in Spain took place at the end of the 18th century, when the laws of the 'confiscation' were enforced. The sale of common lands from 1859 onwards resulted in continuous large-scale clearing of forests and the development of other land uses. In the 1950s the irrigated area increased rapidly, as a consequence of the regulations of river water and the initiation of groundwater extraction (Barberá et al., 1997). This and the beginning of intensification and industrialization of agriculture resulted in massive abandonment of non-mechanisable and marginal areas in the 1950s and 1960s. Initially land abandonment occurred in the most economically developed regions and later throughout the whole country. The area cultivated with cereals decreased drastically, while the area with perennials as olives, almonds and especially citrus increased (Romero Díaz et al., 2002). Land abandonment continued with further economic development, as result of increased importance of tourism and industrialisation. Nowadays intensive agricultural systems are concentrated in the more fertile areas, while marginal areas are under extensive agriculture or have been abandoned (Fernandez Ales et al., 1992). Agricultural land abandonment is expected to increase in the future in many areas of the Mediterranean, as a consequence of changing EU-policies, urbanisation, globalisation, desertification and climate change, which is also projected by different land use change scenarios (Olesen and Bindi, 2002; Rounsevell et al., 2006; Verburg et al., 2006). Most climate change scenarios predict less and more irregular rainfall in the Mediterranean area (Christensen et al., 2007), which will reduce the agricultural productivity in Mediterranean region and lead to increased extensification. However, the recent demand for biofuels might slow down the increase of land abandonment (Hoogwijk et al., 2005). Nevertheless, semi-arid areas like the Carcavo basin probably remain too dry for economically profitable production of crops for food or biofuels.

When abandoned agricultural land is no longer influenced by human activities the secondary succession can start. The development of the vegetation succession depends much on local environmental and ecological factors, especially rainfall and seed dispersal are limiting factors (Pugnaire et al., 2006). In general annual plants and short-lived perennials are dominant during the first phase of abandonment (3-5 years) with a higher cover and species richness (Obando, 2002). In the second phase forbs and dwarf scrubs appear, while perennial grasses and shrubs start to increase after 10 years of abandonment

(Bonet, 2004). *Quercus ilex* dominated shrublands (*Rhamno lycioidis-Quercetum cocciferae*) are considered to be the terminal point of secondary succession in extensive areas of the Mediterranean (Romero-Calcerrada and Perry, 2004). However, in many degraded areas this final stage will not be reached and *Stipa tenecissima* and dwarf shrub communities form the main vegetation on these long abandoned fields. Biodiversity in Mediterranean ecosystems is generally in decline as a consequence of land abandonment in marginal areas and intensification of agriculture and forestry in other areas (Romero-Calcerrada and Perry, 2004; Moreira and Russo, 2007). Reestablishment of natural shrubland in revegetation programs for abandoned agricultural lands has therefore been encouraged by the agricultural policies of the European Union as a means for regenerating the biodiversity of these areas (Caravaca et al., 2003).

The changing vegetation cover and composition after abandonment will also affect soil properties. In general soil quality will improve as a result of the positive influence of vegetation. Studies showed that important soil properties such as organic matter content and aggregate stability increase with time of abandonment (Cammeraat and Imeson, 1998; Kosmas et al., 2000; Dunjó et al., 2003). The recovery of vegetation and improved soil properties will make the soil less vulnerable to erosion. However, the rate of recovery is highly dependent on the amount of rainfall. In more humid parts of Spain a decrease in erosion after land abandonment is observed (García-Ruiz et al., 1996; Molinillo et al., 1997), while erosion in semi-arid areas such as Southeast Spain increased during the first years/decades after abandonment (Cerdà, 1997; Bull et al., 2000; Lasanta et al., 2000). This increase in soil erosion during the first years can be explained by the deteriorated soil properties and still low vegetation cover. Furthermore, soil and water conservation structures are no longer maintained and the original drainage pattern might be restored, which might lead to terrace failure and gully erosion (Gallart et al., 1994; Cammeraat et al., 2005).

1.3. Objectives and research questions

The general objective of this thesis is to study the interactions between soil, vegetation and erosion in the context of agricultural land abandonment at multiple scales in a semi-arid environment. This objective combines the three central themes of this thesis, i.e. soil erosion, scale issues and agricultural land abandonment. To connect the different chapters and to focus on the central themes of this thesis three main key research questions have been formulated.

1. Where does agricultural land abandonment occur and how do vegetation and soil properties change after abandonment?
2. Which are the main soil erosion processes on abandoned land and how to mitigate them?
3. How to integrate plot and hillslope scale influences in runoff and erosion modelling at catchment scale?

Besides the general objective and research questions each chapter deals with a separate topic related to one or more of the central themes. Together these chapters contribute to the main objective and research questions of this thesis. The following objectives were defined for each chapter:

Chapter 2:	To identify vulnerable areas for gully erosion using different scenarios of land abandonment.
Chapter 3:	To investigate the development of spatial heterogeneity in vegetation and soil properties after land abandonment.
Chapter 4:	To assess the extent and causes of erosion and terrace failure on abandoned fields and to discuss options for mitigation.
Chapter 5:	To evaluate which vegetation index is most suitable for upscaling fractional vegetation cover in a semi-arid environment using a high resolution QuickBird image.
Chapter 6:	To model runoff and erosion for a semi-arid catchment using a multi-scale approach based on hydrological connectivity.

1.4. Study area description

The Carcavo basin in Southeast Spain was used as study area, since this catchment was selected as study site in Southeast Spain for the RECONDES project. This catchment fulfilled the various criteria that were established for identification of suitable catchments, which were a moderate size of circa 25 km^2, variation in land use, presence of channels, dominance in marl bedrock, availability of some base data and prior knowledge and reasonably accessible.

1.4.1. Topographical setting

The Carcavo basin is located about 40 km northwest of the city of Murcia in Southeast Spain, near the town of Cieza (38°13′ N; 1°31′ W). It is a third order catchment of 30 km^2 with altitudes ranging from 220 to 850 meter above sea level (Figure 1.2). The basin is characterised by two steep mountain ridges, the Sierra del Almorchon in the northwest and the Sierra del Oro in the eastern part of the catchment. The outlet of the basin drains directly into the Segura River, the major river system of southeast Spain. The large difference in base level between the Segura River and the Carcavo catchment is an important driver for erosive processes, which lead to deeply incised channels in the catchment. Indeed "Carcavo" in Spanish means gully.

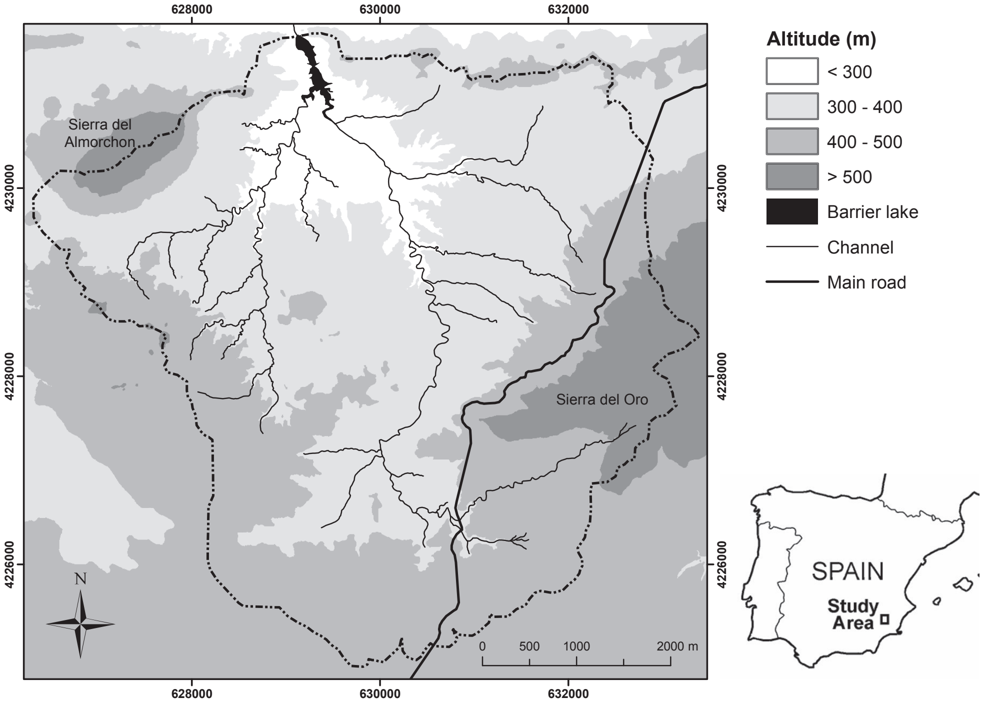


Figure 1.2. Location of the Carcavo basin and its altitude

1.4.2. Climate

The Carcavo Basin is located in the rain shadow of the Betic ranges and this region forms one of the driest areas of Europe. Climate is predominantly Mediterranean semi-arid and ecosystems have to cope with water stress. Average yearly rainfall is just around 300 mm, while the potential evapotranspiration, as measured by the Thornthwaite method, is close to 900 mm. However, the variability between years is very high with a minimum of 125 mm and a maximum of almost 700 mm (Figure 1.3), as measured at the Almadenes weather station, which is located just north of the Carcavo basin. The average annual temperature is 16.5ºC with January being the coldest month with an average temperature of 9.5ºC and July and August the hottest with average temperatures near 26ºC. Frost is relatively uncommon and rarely severe. Intra-annual droughts, typical of Mediterranean climate, are severe. In July and most of August there is virtually no rain, and extended periods of 5 months or more of very low or no rainfall are common. The rainfall regime is bimodal with most rainfall in April and October. Especially autumn rains can be very intense, when the Mediterranean Sea is still warm and humid air from the east can lead to a very unstable atmosphere due to convective and orographic effects. Under these conditions very intense rainstorms (*gota fria*) can locally from that do most of the geomorphological work (Poesen and Hooke, 1997). In October 1973 the nearby Guadalentín basin was struck by such an extreme event, with 350 mm of rain within 10 hours, which resulted in a flash flood that killed 96 people. In various vulnerable catchments dams and reservoirs were constructed to reduce the risk on flash flood in downstream areas. A barrier dam was also constructed at the outlet of the Carcavo basin in the late eighties.

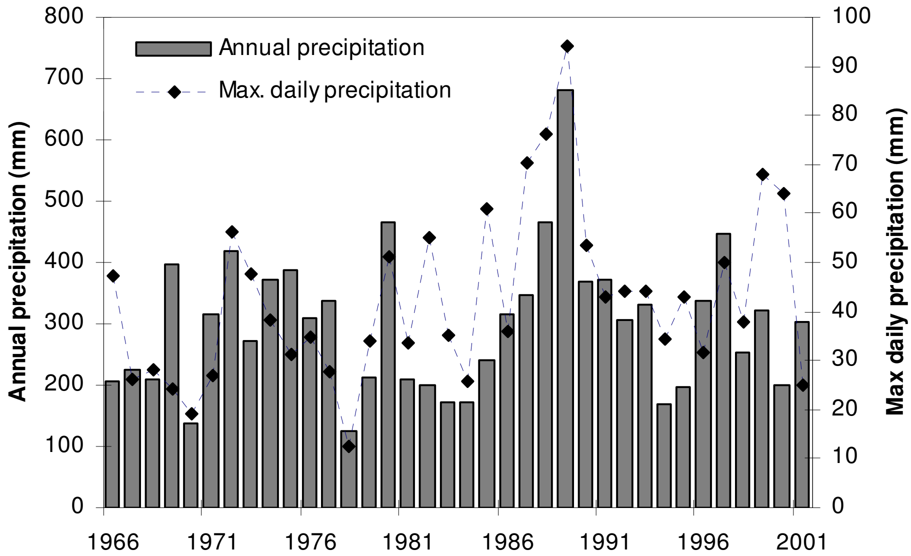


Figure 1.3. Annual and daily maximum precipitation for the Almadenes weather station

1.4.3. Geology and geomorphology

The Carcavo basin is located in a complex geological setting within the external zone of the Betic Cordillera. Two important fault systems border the catchment, at the northern border the Linea Electrica fault, which is an east to west oriented strike-slip fault and the Crevillente fault south of the catchment, which is a southwest to northeast oriented strike-slip fault. The Crevillente fault system divides the internal and the external zone (Biermann, 1995; Nieto and Rey, 2004). The internal zone is situated south of this fault system and consists mainly of metamorphic rocks, whereas the external zone is situated north of the fault zone and consists of sedimentary rocks. The zone between the Crevillente fault and the Linea Electrica fault is the Subbetic zone to which the Carcavo basin belongs, consequently calcareous sedimentary rocks are found.

During the late Burdigalian (early to middle Miocene) the oblique collision of the African plate against Iberia caused strike-slip deformation. This resulted in basin subsidence and uplift of independent units in the fault zone. The Carcavo basin was one of the basins that subsided, which lead to the formation of marine marls, sandstones and conglomerates during the Tortonian. These marls are currently folded and deformed due to Pliocene tectonic activity along the main faults. Also diapirism of the plastic underlying Keuper marls caused deformation and subsidence of the Tortonian marls. At present the resistant sandstone and conglomerate ridges are located along the edges of the Tortonian substrate. The southern part of the catchment was in the past not connected to the northern part and drained to the south. During the Late Pleistocene this part became connected due to backward erosion of the Carcavo channel. The capturing of the southern part could be deduced from the sudden supply of large amounts of distal fine material to an extensive area in the northern part and the disconnection between the Middle Pleistocene colluvial deposits at the southern footslopes of the Sierra del Oro (Van Gorp, 2006). During the Quaternary continuing neo-tectonic activity and sea levels changes lead to lowering of the Segura river bed (Faust, 1997). In response to this base level lowering the Carcavo channel started to incise and especially the northern part of the catchment is now dominated by incision patterns, whereas the incision in the southern part is not so strongly expressed, due to the relatively late connecting to the main Carcavo basin. At present, the catchment is still reacting to the last base level lowering, which causes incision and mass movement in the entire Carcavo basin.

In the Carcavo basin five main geological units can be distinguished. The oldest are of Triassic age consisting of Muschelkalk carbonate rocks and highly deformed gypsiferious Keuper marls. The highest parts of the basin consist of Jurassic limestones and dolomites, which are the Sierra del Almorchon in the northwest and the Sierra del Oro in the eastern part of the catchment. In the southeast of the basin deformed marls, limestone and sandstone of Cretaceous age are the main lithology. The centre of the basin consists of Miocene basin fills, which are mainly marls and some sandstones and conglomerates. The last geological unit that can be distinguished are the Quaternary slope deposits, which overly the Miocene and Cretaceous formations. In these Quaternary deposits several surface levels can be distinguished, which represent the different periods of geomorphological activity. Van Gorp (2006) mapped four surface levels at the footslopes of the Sierra del Oro and Sierra del Almorchon, and two terrace levels along the Carcavo channel.

Within the Carcavo basin several well developed pediments are present, of which the highest two levels are covered by calcretes. Pediments are slightly sloping areas, mainly developed in bedrock or covered with a thin covering of colluvial materials on top and connected to the steeper headwalls of hills and mountains. These pediments are characteristic for long term landscape development in semi-arid environments (Cooke et al., 1993). Pediments covered by calcretes are stable landforms due to the protective caprock function of the calcrete (Alonso-Zarza et al., 1998). Calcretes are pedogenetic horizons that are cemented by calcium carbonate. In the semi-arid area of southeast Spain many calcretes are present, which have been formed during different stages of the Pleistocene. Under semi-arid to semi-humid conditions (300-600 mm rainfall) soil formation occurs and a petrocalcic horizon can develop in the subsoil. When the climate becomes more arid, the top layer denudates and the petrocalcic horizon becomes exposed to the surface and protects the underlying substrate against further erosion (Blümel, 1986).

1.4.4. Soils and soil properties

In general soils in the Mediterranean on relatively stable surfaces are characterised by a large proportion of limestone and other calcareous rocks as parent material, moderate weathering, hematite-induced reddening of clays and carbonate dissolution and reprecipitation with prevalence of calcic horizons (Yaalon, 1997). However, within the Carcavo basin four main soil types occur, these are Leptosols on the limestone and dolomite mountains and outcrops, Calcisols on the pediments of the Sierra del Almorchon and Sierra del Oro, Regosols on the marls and Gypsisols on the gypsiferious Keuper marls. Soil formation is limited due to the semi-arid climate and the steep topography with subsequent high erosion rates, resulting in mostly shallow soils, i.e. Leptosols and Regosols. As a consequence lithology is the main soil forming factor and the distribution of soil types is strongly linked to it. Soils in the Mediterranean are often susceptible to water erosion, due to low organic matter content, large silt fraction, poor soil structure and weak aggregate stability, which leads to surface sealing during intense rainfall (Ramos et al., 2000). These soil crusts reduce infiltration rates and may increase runoff and consequently erosion. Significant differences in erodibility are found among lithologies with marls as most erodible material (Albaladejo et al., 1995; Cerdà, 1999). Marl is also one of the main substrates in the Carcavo basin with an occurrence of 60 percent, and even 77 percent when the gypsiferious Keuper marls are included, which makes the catchment in combination with the sparse vegetation cover vulnerable to erosion.

The spatial distribution of soil properties in semi-natural areas is often very heterogeneous under semi-arid conditions due to patchiness of the vegetation. The higher organic matter input by litter favours soil aggregation and soil faunal activity, which both increase macro-porosity and infiltration rates and lower surface compaction. Also the micro-climate under vegetation with lower temperature amplitudes and more shading is favourable, resulting in better soil moisture conditions than for bare soil (Bochet et al., 1999; Maestre and Cortina, 2002). All these characteristics lower the soil erodibility and decrease the erosion risk. Under vegetation also other chemical soil properties such as nutrient content and cation exchange capacity increases. The increased heterogeneity of soil resources is also known as the “islands of fertility” phenomenon (Schlesinger et al., 1990). Organic carbon content is generally the best indication for soil quality because of its influence on soil structure, water holding capacity and CEC, since most soils in the Mediterranean are often water limited instead of nutrient limited.

Besides the chemical and physical properties of the soil, several other factors such as rock fragments, biological soil crusts and water repellency, influence the hydrological and erosion response of a soil as well. The presence of rock fragments in the soil has significant effects on the infiltration and the water storage capacity of the soil as well as on the soil surface. As rock fragments are usually present at the surface it is important to notice whether these are positioned at the surface or are embedded in the soil surface crust. In the latter case infiltration is reduced, whereas in the first case infiltration is increased (Poesen and Lavee, 1994). Biological or cryptogamic soil crusts, which are thin crusts made up of mosses, lichens, algae, and bacteria, are another typical feature in semi-arid ecosystems. These organisms can form a resistant biotic layer in bare areas in undisturbed arid and semi-arid lands. They form a cryptogamic crust that helps to protect soil material from erosion, absorbs moisture, and provides nitrogen and other nutrients for plant growth (Harper and Marble, 1988; Belnap, 2006). Another property of many Mediterranean soils, especially coarse textured soils, is their water repellency, which has substantial hydrological and geomorphological repercussions. Reduced infiltration capacity, enhanced overland flow, accelerated soil erosion and development of preferential flow paths in the soil are the main effects. Hydrophobicity is caused by organic coatings of long-chained organic molecules, released from decomposing or burning plant litter, micro-organisms, fungal growth or root zone and leaf surfaces of living plants (Doerr et al., 2000). It is shown that several Mediterranean vegetation species commonly occurring in the study area have different degrees of hydrophobicity (Verheijen and Cammeraat, 2007).

1.4.5. Land use

Due to its topographical setting and the lack of human habitation the Carcavo basin is mainly used for extensive agriculture and reforestation. The current land use in the study area consists of cereals, olive and almond orchards, vineyards, abandoned land, reforested land and semi-natural vegetation. The north slopes of the Sierra del Oro and Sierra del Almorchon, which receive less radiation, can sustain a semi-natural pine forest, while other semi-natural areas consist of shrublands and *Stipa tenecissima* dominated rangelands.

Agriculture is mainly situated on the plains, wide streambeds and terraced hillslopes. Within the Carcavo basin few houses are located, but most of them have been abandoned and are now ruins. Livestock is only a marginal agricultural activity with two small mixed sheep and goat flocks that pass the area now and then to graze on marginal agriculture fields and semi-natural areas. Close to the outlet of the catchment a barrier dam was built in the late eighties for the purpose of flood control. Irrigation possibilities within the catchment are limited and only few water basins have been constructed during the last years. These are fed by local wells and groundwater and are mainly used for olives. However, some almond orchards are occasionally irrigated as well. Just north of the basin, in the valley of the Segura river, intensive irrigation systems exists for peach orchards.

Three different stages in the agricultural history of the Carcavo basin can be distinguished. The first half of the twentieth century was characterized by an extension of traditional dryland crops and livestock in the catchment. From 1950s to 1980s agriculture suffered and there was large scale land abandonment due to general socio-economic factors relating to the marginal areas in Spain (Fernandez-Ales et al., 1992; Barberá et al., 1997). Strong migration from the rural areas to the cities took place at the same time as government reforestation projects tried to recover abandoned land. In the Carcavo basin large parts of degraded rangeland were reforested with pine (*Pinus halepensis*) during the 1970s for reforestation and soil conservation purposes. Nowadays these reforested lands make up almost 45 percent of the total area, while croplands occupy about 29 percent. The last stage of land use change is the conversion of rainfed cereals to almonds, olives and vineyards since the eighties. Especially almond orchards, which are more drought tolerant, have replaced cereals and are now dominant in the southeastern part of the catchment. In addition, parts of the non-irrigated agriculture have been abandoned during the last decades and are now under different stages of secondary vegetation succession.

1.4.6. Vegetation

The flora of Southeast Spain is characterised by an exceptional richness and is very abundant in endemic and Iberian-Mauritanian species (Peinado et al., 1992). Within the Murcia Region 292 associations and communities are described, which is higher than many Spanish or European similarly sized territories (Alcaraz et al., 2000). The natural vegetation on slopes is mainly composed of *Stipa tenacissima* communities and dwarf-shrubs with *Rosmarinus officinalis*, *Cistus clusii*, *Thymus membranaceus* on mid and low areas. Harvesting of *Stipa tenacissima* tussocks was an important economic activity during the last centuries, which favoured its widespread occurrence in this part of Southeast Spain (Yanes, 1993). Slope vegetation at higher altitudes comprises *Rhamno lycioidis-Quercetum cocciferae* shrublands with *Juniperus oxycedrus* and *Pistacia lentiscus*, which is considered the climax vegetation in this semi-arid climate. On the Keuper formation, which is rich in gypsum, several endemic scrub species such as *Ononis tridentata*, *Salsola genistoides*, *Teucrium carolipaui* and *Helianthemum squamatum* are common. Vegetation in channels and gullies is characterized by patches of riparian and salt tolerant semi-arid vegetation, with species such as *Nerium oleander*, *Tamarix canariensis*, *Brachypodium retusum*, *Juncus acutus* and reed beds of *Phragmites australis* (Navarro Cano, 2004). In reforested areas *Pinus halepensis* is the dominant species, followed by *Stipa tenacissima*, *Brachypodium retusum* and *Rosmarinus officinalis*, other species are insignificant in terms of cover.

Although this general vegetation description gives a short overview of the plant communities in the study area, the local conditions finally determine which vegetation species will occur. Especially slope exposure has a large effect on vegetation in semi-arid areas with high relief (Guerrero-Campo et al., 1999). North-facing slopes can sustain a higher vegetation cover and vegetation succession is faster due to lower evapotranspiration. On the northern slopes of the Sierra del Oro and Sierra de Almorchon a semi-natural forest is growing, while the southern slopes are mostly bare.

Plant community dynamics of Mediterranean basin ecosystems are mainly driven by an alternation of periods of human intervention and land abandonment. As a result, a mosaic of plant communities has evolved following different stages of degradation and regeneration (Gallego Fernández et al., 2004). Within the Carcavo basin abandoned fields with different stages of secondary vegetation succession are found. During the first years of abandonment the vegetation is dominated by annuals. Later dwarf shrub species and grasses become more important, *Artemisia herba-alba* and *Plantago albicans* are typical species for these stages of abandonment. The final stage of secondary vegetation succession is formed by *Rosmarinus officinalis* and *Stipa tenacissima* dominated communities. An ultimate successional stage of forest vegetation with *Quercus ilex* is not likely to occur under the semi-arid conditions of Southeast Spain, due to low water resources and intensive human disturbances over millennia (Rivas-Martínez, 1987).

1.5. Thesis outline

This thesis consists of seven chapters, of which this introductory chapter is the first one. Chapter 2 to 6 are based on scientific papers that have been published or have been submitted to peer reviewed international journals. In Chapter 2 vulnerable areas for gully erosion are identified under different scenarios of land abandonment for the Carcavo basin. A field survey revealed that abandoned land had more gully erosion compared to cultivated land. With a spatially explicit land use change model four scenarios with different rates of land abandonment were simulated and potentially vulnerable areas for gully erosion were identified. Chapter 3 zooms in to the plot scale and describes the development of spatial heterogeneity in vegetation and soil properties after land abandonment. For two series of abandoned fields the vegetation succession, soil properties and vegetation patterns from detailed aerial photographs were analysed. Chapter 4 deals with a field scale study of the factors that increase erosion risk on abandoned fields. Since terrace failure was one of the major soil erosion processes on abandoned fields, we studied the causes of terrace failure in more detail. This chapter also discusses potential soil and water conservation practices for the mitigation of soil erosion after land abandonment. Chapter 5 describes the upscaling of fractional vegetation cover based on detailed aerial photographs and a QuickBird satellite image for the entire catchment. Different vegetation indices were evaluated for their suitability of upscaling fractional vegetation cover in a semi-arid environment. In Chapter 6 data and results from the previous chapters are integrated in a catchment scale runoff and erosion model. Vegetation patches at plot scale and agricultural terraces at hillslope scale were the most important sinks for runoff and sediment. The influences of these sinks were quantified and integrated in the infiltration module of the LAPSUS model to simulate the effect on the hydrological connectivity. Finally, a synthesis is given in Chapter 7, which summarises the main results in relation to the three central themes and discusses the implications of this thesis.

2. Identification of vulnerable areas for gully erosion under different scenarios of land abandonment in Southeast Spain *

2.1. Introduction

Due to changing European policies, urbanisation, globalisation, desertification and climate change land abandonment has become one of the main changes in land use in Mediterranean countries (Geeson et al., 2002). Land abandonment may have large consequences on erosion and sedimentation processes. On the one hand, succession of semi-natural vegetation after abandonment might increase the vegetation cover, improve soil properties, and decrease runoff and erosion (García-Ruiz et al., 1996; Molinillo et al., 1997). On the other hand, soil and water conservation structures, such as terraces, might collapse due to lack of maintenance and piping and consequently increase erosion (Gallart et al., 1994; Faulkner et al., 2003). Pardini et al. (2003) suggest that especially during the first years after abandonment runoff and erosion occur. Ruiz-Flaño et al. (1992) describe for the Spanish Pyrenees that also the presence of grazing is of importance, since grazing implies periodic burning, which increases surface stoniness and overland flow. The consequences of land abandonment for runoff and erosion are largely dependent on the specific characteristics of the location of land abandonment. Therefore, to mitigate runoff and erosion from abandoned land, it is necessary to know more about where land abandonment will most likely occur.

During the last decade many land use change models have been developed to simulate deforestation, urbanisation and intensification of agriculture (Briassoulis, 2000; Verburg et al., 2004). Land abandonment, however, has rarely been addressed in land use modelling studies (Mulligan, 2004), despite the fact that it is one of the most widespread changes in land use in southern Europe and the potentially large impacts on different environmental and socio-economic processes. To develop a model for land abandonment, we first need to understand the driving factors that lead to abandonment. According to MacDonald et al. (2000) land abandonment is the result of intensification of agriculture, technological developments, and the influence of Common Agricultural Policy. This increased productivity and focused agricultural activities on more fertile and accessible land, while

* Published as Lesschen, J.P., Kok, K., Verburg, P.H. and Cammeraat, L.H. 2007. Identification of vulnerable areas for gully erosion under different scenarios of land abandonment in Southeast Spain. Catena 71: 110-121.

marginal lands were abandoned. Other factors influencing land abandonment are industrialisation and increased economic importance of tourism (Fernandez-Ales et al., 1992). However, each location has specific factors determining the rate and pattern of land abandonment. Especially, the influence of biophysical factors on land abandonment is still largely unknown in quantitative and spatial terms.

One of the environmental consequences of land abandonment is land degradation and gully erosion in specific. Recent studies indicate that gully erosion is one of the main erosion processes in terms of soil losses and sediment production (Poesen and Valentin, 2003). As Martínez-Casasnovas et al. (2003) calculated an average net erosion of 576 ton ha^{-1} yr^{-1} for a gully area in Northeast Spain. Plata Bedmar et al. (1997) concluded on the basis of Cs_{137} research that 60 percent of the sediments in the Puentes reservoir (Southeast Spain) came from subsurface soil horizons, for which gully and river channel erosion could be held responsible. And a survey by Verstraeten et al. (2003) within the catchments of 22 Spanish reservoirs clearly indicated that the sediment yield increases when the frequency of gullies increases in the catchment. Gully erosion is therefore an important soil degradation process (Poesen et al., 2002), and these gullies form effective links for transferring runoff and sediment from uplands to valley bottoms and permanent channels (Poesen et al., 2003).

Many cultivated and abandoned fields in Southeast Spain are located on marls, which are very susceptible to slaking and crusting. By absence of tillage the crust starts to develop and will decrease infiltration, which causes an increase of runoff and triggers the formation of rills, which will develop into gullies (Imeson et al., 1998). Furthermore, fields on slopes are often terraced and after abandonment the original drainage pattern will restore, leading to terrace failure. We expected therefore that in these semi-arid areas abandoned land is more vulnerable to gully erosion than cultivated land, due to the absence of frequent tillage, lack of maintenance of terraces and earth dams, and low vegetation cover during the first years after abandonment.

The objective of our study was to identify vulnerable areas for gully erosion using different scenarios of land abandonment in Southeast Spain. We selected the Carcavo basin in Southeast Spain as our study area, as this region is representative for marginal agricultural land in semi-arid areas, and land abandonment is the main change in land use over the last decades. With a preliminary field survey we checked whether abandoned land indeed is more vulnerable to gully erosion than cultivated land. Subsequently we simulated the spatial dynamics of land abandonment in the study area with a spatially explicit land use change model for four different land use change scenarios. These results were used to identify vulnerable areas for gully erosion due to land abandonment by a simple GIS-model based on the controlling factors of gully erosion.

2.2. Study area

The Carcavo basin is located about 40 km northwest of the city of Murcia in Southeast Spain, near the town of Cieza. Its UTM coordinates are 4228000 m N and 630000 m W (European_1950 datum zone 30N). It is a small catchment of 30 km^2 and altitudes range between 220 and 850 meter. This region of Spain is very dry with an average annual rainfall of 300 mm and a potential evapotranspiration of 900 mm. The geology of the area consists of steep Jurassic limestone and dolomite mountains with calcareous piedmonts, basin deposits of Cretaceous and Miocene marls, and Keuper gypsum deposits (Figure 2.1). Most soils in the area are thin (Leptosols), weakly developed (Regosols) and mainly characterised by their parent rock (Calcisols and Gypsisols).

The current land use in the study area consists of cereals, grapes, olives, almonds, abandoned land, reforested land and semi-natural vegetation (see Figure 2.3 for a current land use map). In the 1970s large parts of degraded land were reforested with pine (*Pinus halepensis* Mill.) within the framework of reforestation and soil conservation programs. Some almond and olive fields in the central part are under irrigation, while low-yielding cereals are found on marls without irrigation. During the last decades, parts of the non-irrigated agriculture have been abandoned and are under different stages of secondary succession. The steeper and higher areas are under semi-natural vegetation and on north slopes under forest.

2.3. Methods

2.3.1. Gully survey

With a preliminary field survey we compared abandoned sites with similar cultivated sites to verify the assumption that abandoned land is more vulnerable to gully erosion than cultivated land. In the Carcavo basin we selected six field sites (Figure 2.1), an abandoned site versus a cultivated site, which had the same lithology and topographic position. The survey took place in June 2005 and from each gully at the site the coordinates were taken and a description was made. The following characteristics were included: gully activity (three classes), vegetation cover in the gully (three classes), type of gully head according to the classification of Oostwoud Wijdenes et al. (1999), size of the gully, estimation of the volume, slope, and presence of a plunge pool. Site 1 and 2 were located on a plateau of Miocene marl with Quaternary slope deposits and surrounded by deep channels, which resulted in the occurrence of mainly bank gullies. Site 3 and 4 were situated in a wide valley in Miocene marl and gullies occurred mainly in terrace walls. Finally, Site 5 and 6 were located around a channel head in Cretaceous marl and both bank gullies and gullies in terrace walls were found. The three abandoned sites were before all under cereals and have been abandoned 10-20 years ago.

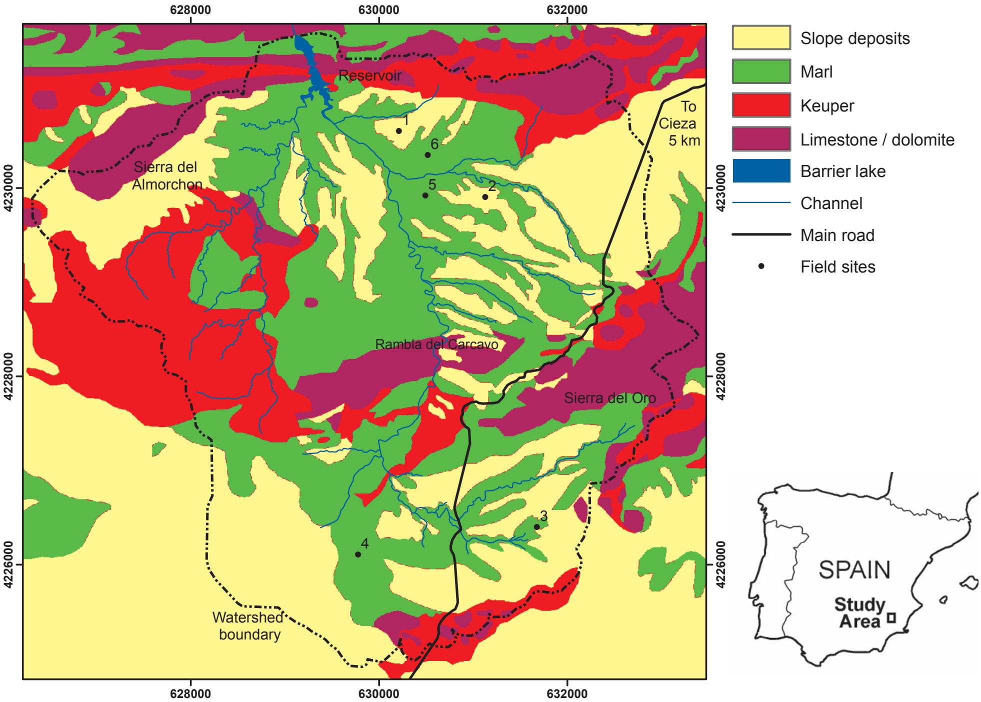


Figure 2.1. Geological map of the Carcavo basin with the location of the field sites

2.3.2. CLUE-s model

We used the CLUE-s (Conversion of Land Use and its Effects version 2.3) model, which is specifically designed to address the spatial dynamics of land use change. It is a model that has been tested in many different environments and proven capable to adequately simulate changes in land use pattern (Verburg et al., 2002; Verburg and Veldkamp, 2004). The advantages of the CLUE-s model are the explicit attention for the functioning of the land use system as a whole, the capability to simulate different land use types at the same time and the possibility to simulate different scenarios. The model is based on an empirical analysis of location suitability for each land use combined with a dynamic simulation of competition and interaction between the spatial and temporal dynamics of land use. The model allocates predefined land use requirements, determined at the regional level, to grid cells in an iterative procedure that includes location characteristics, spatial policies and restrictions, and land use type-specific conversion settings. The following paragraphs describe: (1) the calculation of driving factors, (2) the definition of four land use change scenarios, and (3) the allocation procedure of the CLUE-s model.

2.3.2.1. Driving factors

Policy relevant approaches to assess land use change must be based upon a number of multi-disciplinary mechanisms, since the landscape is a complex system of interacting human, physical and natural processes that create unique landscapes that change through time (Oxley and Lemon, 2003). These interacting processes are driven by one or more variables that influence the decisions of land users. These variables are often referred to as underlying driving forces that underpin the causes of land use change (Geist and Lambin, 2003). Especially biophysical factors, e.g. slope or soil type, pose constraints to land use change at certain locations. For the study area, we prepared a list of assumed driving factors, based on common theories on driving factors of land use change and knowledge of the conditions in the study area. The spatial variation of these supposed driving factors was quantified based on different maps and aerial photographs (Table 2.1). All these data were converted into grids with a cell size of 25 meter. The relations between current land use and the spatial variation in the driving factors were thereafter evaluated using logistic regression analysis. The logistic regression indicates the probability of a certain grid cell to be devoted to a land use type given the values of the driving factors at that location. The coefficients were estimated using the current land use pattern as dependent variable. The resulting logistic regression coefficients were standardized according to Menard (2001), which allows the comparison of the strength of the relationship between the dependent variable and different independent variables.

Table 2.1. Description of available data sets

Name	Description
Land use	Land use maps derived from aerial photographs of 1985 and from a field survey in 2004
Altitude	Digital Elevation Model (DEM) with a resolution of 5 m, based on 1:5000 contour lines
Slope	Derived from DEM
Potential radiation	Derived from DEM, according to McCune and Keon (2002)
Topographic position	Derived from DEM (classes: flat, valley, slope and top) with ArcView script of Jenness (2005)
Lithology	Geological maps at scale of 1:50,000 (IGME, 1972, 1982), which serve as a proxy for soil type
Housing density	Interpolated map of houses, derived from 1:25,000 topographic maps (CNIG, 1999, 2003)
Irrigated area	Map based on land use, field observations and distance to water basins
Accessibility	Interpolated distance to road (derived from topographic maps)
Travel time to town	Travel time to the town of Cieza, calculated with the cost allocation procedure in ArcGIS, which takes account of road quality
Wetness index	Derived from DEM, according to Beven and Kirkby (1979)

Stepwise procedures combined with expert knowledge were used to reduce collinearity and to select the factors with a relevant and supposed causal relation to the spatial distribution of land use. Logistic regression assumes the data to be statistically independent and identically distributed. However, spatial land use data have the tendency to be dependent, known as spatial autocorrelation (Anselin and Griffith, 1988). We minimized the influence of spatial autocorrelation on the regression by performing the regression on a random sample of grid cells at a certain minimum distance from each other (Overmars et al., 2003). To evaluate the performance of the regression model we used the Relative Operating Characteristic (ROC), which is a common measure for the goodness-of-fit of a logistic regression model. The ROC of an ideal model outcome is 1, while a random classifier achieves approximately 0.5 (Swets, 1988).

2.3.2.2. Land use change scenarios

Since Roman times humans have altered the land in the Mediterranean area. Clearing, reclamation and terracing of large areas in the inland took place mainly in the 17th and 18th century as a consequence of high population pressure (Ruecker et al., 1998). In the 1950s intensification and industrialisation of agriculture led to expansion of irrigation and also to massive abandonment of non-mechanisable areas (Barberá et al., 1997). Intensive agricultural systems, mainly fruit trees, began to concentrate in the more fertile areas, while marginal areas, mainly cultivated with cereals, were abandoned (Romero Díaz et al., 2002). During the last decades, abandonment continued, as a result of industrialisation and increased economic importance of tourism (Fernandez-Ales et al., 1992). In the Carcavo basin land use changed considerably, especially in the seventies due to the large scale reforestation. Furthermore a large shift in agricultural land use occurred, as almonds, olives and grapes replaced cereals. Abandonment of agricultural land will probably increase in many regions of the Mediterranean because of changing socio-economic and climatic circumstances. Decreasing EU-support, continuing rural-urban migration, increasing production costs, and increasing land degradation will lead to abandonment. Furthermore, most climate change scenarios predict less and more irregular rainfall in the Mediterranean (Christensen et al., 2007). This will reduce agricultural productivity and lead to further abandonment (Olesen and Bindi, 2002).

We used the scenarios from the MedAction project (De Groot and Rotmans, 2004), because these were adapted towards rural and more isolated areas in the Mediterranean and developed to serve local decision-making with regard to policy formulation for sustainable land management to combat desertification. The three scenarios are *Knowledge is King* (KK), *Big is Beautiful* (BB) and *Convulsive Change* (CC) (Kok et al., 2006). In the KK-scenario ICT revolution and important inventions trigger fundamental societal changes. Initially agriculture experiences large problems with increased competition from the east when the EU expands. Later, the position of the entire agricultural sector improves strongly, when the water availability problem is solved and irrigated agriculture expands. The tourist industry becomes the most important economic sector in the south of Spain, benefiting from cheaper transport, increased water availability, and Sunbelt formation. More investments in reforestation programs speed up the expansion of forested areas. In the BB-scenario a widespread 'merger mania' oversteps all limits, initiating societal degeneration. First, increased competition, radical reforms of the Common Agricultural Policy and phasing out of most subsidy systems are a severe blow especially for the smaller farmers. Later climate change initiates the total collapse of agriculture, leading to widespread land abandonment. Finally, in the CC-scenario accelerated climate change triggers desert formation and outpaces society's ability to adapt. Decreasing precipitation and water availability terminate most rainfed agriculture. Irrigation is first strongly reduced, but later the surviving farmers profit from the water transportation network. Towards 2030, society gradually shows signs of resilience and slowly learns how to deal with droughts, water scarcity in cities and in the tourist and agricultural sectors, as well as the resulting land degradation. The scenarios are described in Table 2.2, and have a time frame until 2030.

Table 2.2. Summary of the changes of factors, actors and sectors in the three scenarios (Kok et al., 2006)

	Convulsive Change	*Knowledge is King*	*Big is Beautiful*
Factors			
Water availability	Decreases	Increases strongly	Decreases
Land degradation	Increases strongly	Largely controlled	Increases
Migration	Increases strongly	Increases very strongly	Increases very strongly
Economic stability	Decreases (strongly)	Relatively high	Decreases very strongly
Sectors			
Agriculture	Severely weakened	Generally strong but divided	Collapses without recovery
Tourism (number)	Decreases slightly	Increases very strongly	Decreases strongly
Forest fires	Increases, but controlled	Increases slightly	Uncontrollable problem
Civic	Generally healthy	Healthy but divided	Unhealthy and divided
Actors			
Gov. bodies (EU)	Relatively small	EU expands greatly	Power supra-national
Businesses	Environmentally friendly	Strong influence	Very powerful, later damaged
NGOs	More important	Highly organised, powerful	Small role
Scientists	Influence small	Key actor	Science stagnates

Based on the MedAction project, which parameterised and quantified the same scenarios for a spatial policy support system for the Guadalentín watershed in Southeast Spain, we estimated the effect of the foreseen changes on the land use requirements for the Carcavo basin for 2004 to 2015 (Figure 2.2). Besides the three described scenarios we used a base scenario that represents the continuation of land use change as observed during the previous period (1985-2004). During the first years the trend of the base scenario is followed and from 2008 the influence of each scenario is incorporated, because current land use policies have already a certain projection period. In the KK-scenario a strong decrease in cereals is projected and an increase of 2 percent per year for almond/olive and grapes as a result of more available water, consequently the area of abandoned land first increases and decreases after 2010. The BB-scenario projects after the first years a strong decrease of 10 percent per year for almond/olive and grapes and 15 percent per year for cereals due to the abolishment of subsidies, which results in an enormous increase of abandoned land. Finally the CC-scenario has a decrease of 3 percent per year for almond/olive and grapes and 10 percent per year for cereals, due to the decreasing water availability. The land use requirements for the other land uses that are not in Figure 2.2 do not or minimally change, only for the KK-scenario an increase of reforested areas is taken into account, because more is invested in reforestation programs for desertified areas.

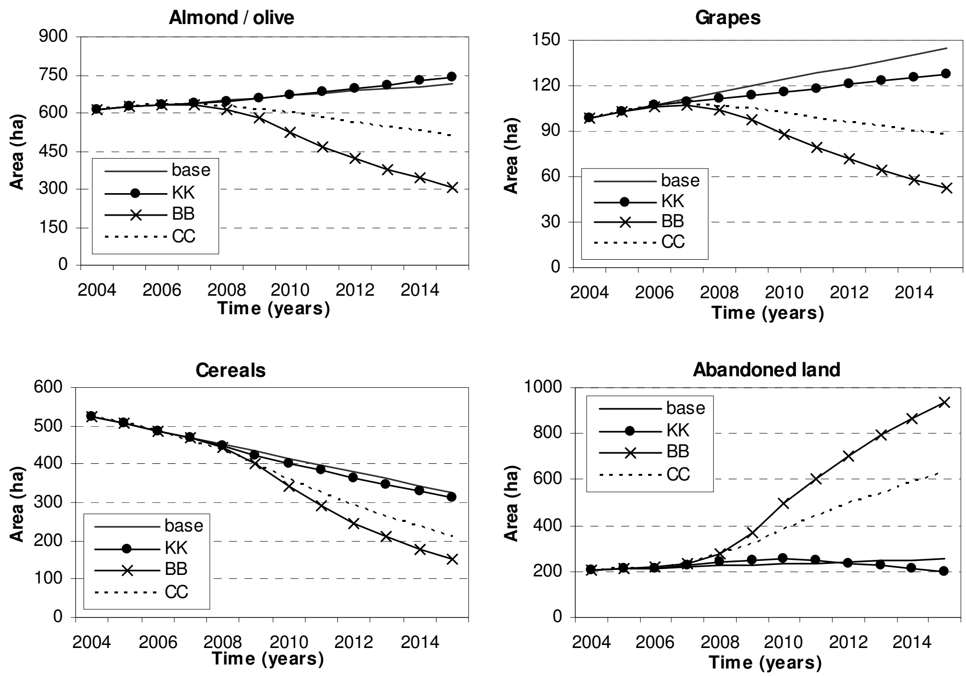


Figure 2.2. Land use requirements for the four scenarios

2.3.2.3. Allocation procedure

The land use change allocation procedure of the CLUE-s model makes use of the derived logistic regression models and the change in land requirements as described above in combination with dynamic modelling of the competition among land uses. Land use type specific conversion settings determine the spatio-temporal dynamics of the simulation. Two sets of parameters characterize the individual land use types: conversion elasticities and land use transition sequences. The conversion elasticities are related to the reversibility of land use change. Land use types with high capital investment will not easily be converted in other uses as long as there is sufficient demand, e.g. olive groves. Other land use types easily shift location when the location becomes more suitable for other land use types, e.g. abandoned land. The land use transition sequences are specified in a land use conversion matrix. For each land use type it is indicated in what other land use types it can change during the next or following time steps. The conversions that are restricted by a certain spatial policy can be indicated in the land use conversion matrix as well, e.g. reforested land and semi-natural vegetation are not allowed to change into agriculture.

When all input is provided the CLUE-s model calculates, with discrete time steps, the most likely changes in land use given the above described restrictions and suitabilities. For all grid cells the total probability is calculated for each of the land use types, based on the location suitabilities following the logistic regressions, the conversion elasticity and an iteration variable that is indicative for the relative competitive strength of the land use type. For each location the land use with the highest total probability is allocated as long as the change does not contradict the settings of the conversion matrix. In an iterative procedure the iteration variable is adjusted until the allocated area equals the land requirements for all land uses.

2.3.2.4. Calibration

We calibrated the model for the period 1985-2004 for which the land use changes were known. The land use conversion elasticities were used as calibration parameters and with the multiple resolution procedure, developed by Costanza (1989), we calculated the model goodness-of-fit. This method yields indices that summarize the proportion of agreement over a range of different resolutions. Since the calculated goodness-of-fit of the model is only a relative value it had to be compared with a certain default, in this case the null model was the map with the initial conditions of 1985. After the calibration the model was run for the period 2004 to 2015 for the four scenarios.

2.3.3. Gully erosion vulnerability model

The final step is the link from abandoned land to vulnerable areas for gully erosion. Several models have been developed for the prediction of gully erosion rates and volumes, however, all these models lack the routine to predict the location of gully erosion (Poesen et al., 1998). Therefore we constructed a simple GIS-model based on the controlling factors of gully erosion to predict the location of potentially vulnerable areas for gully erosion. These controlling factors are soil type, land use, climate and topography (Poesen et al., 2003; Valentin et al., 2005). Since the study area is small the spatial variation in rainfall can be neglected, which leaves three factors that have to be included in the model. The map with the simulated abandoned lands in 2015 served as an input for the land use factor. The geological map served as a proxy for soil type, because soils in these semi-arid areas are closely related to the lithology. All marls and Quaternary fills were considered to be potentially vulnerable to gully erosion, because gullies occur more frequent in soil types with high silt content (Vandekerckhove et al., 2000a).

For the topography factor we used the topographic threshold concept, which predicts locations in the landscape where gully heads might develop. The topographic threshold can be represented by a negative power-equation: $S = aA^{-b}$, where S is the local slope, A the drainage area and a en b coefficients depending on the environmental characteristics. This S-A relation describes the position in the landscape where ephemeral and permanent gully heads may develop (Poesen et al., 2003). For Southeast Spain Vandekerckhove et al. (2000b) found values for a around 0.16 and for b 0.15. With these values we calculated the critical slope and selected the areas where the real slope exceeded the critical slope. Finally we derived the potentially vulnerable areas for gully erosion, which should be under abandoned land, have erodible soils, and be in a topographic position where gully heads might develop.

2.4. Results

2.4.1. Gully survey

Table 2.3 shows the results of the gully survey. All abandoned sites had more gullies and a higher gully density than the comparable cultivated sites. On one of the cultivated sites no gully at all was found. When we compare the estimated gully volumes the difference is even more pronounced. The gully volume on the abandoned sites is about a factor seven higher compared to the cultivated sites, although the variation is very large. Gully activity was higher for most abandoned sites, while vegetation cover in the gully was not significantly different.

Table 2.3. Averaged gully characteristics per site (standard deviation in parentheses)

Site	Land use	Size (ha)	Position	Gullies (number)	Density (gully/ha)	Activity*	Vegetation*	Volume (m^3)
1	Abandoned	17	Plateau	18	1.1	1.7 (0.7)	1.0 (0.6)	55 (101)
2	Almond	11	Plateau	5	0.5	0.6 (0.5)	1.2 (0.4)	7.7 (9.2)
3	Abandoned	1.4	Channel head	18	12.6	1.8 (0.4)	0.9 (0.7)	37 (73)
4	Almond	3.4	Channel head	7	2.1	1.9 (0.4)	0.7 (0.5)	4.9 (6.8)
5	Abandoned	4.4	Valley	11	2.5	1.6 (0.8)	0.8 (0.6)	2.2 (4.0)
6	Almond	4.2	Valley	0	0	-	-	-

* Activity and vegetation are ranging from low (1) to high (3)

2.4.2. Land abandonment modelling

Table 2.4 shows the driving factors and the standardized beta-coefficients for each land use type for 2004. Negative values mean that the probability will decrease upon an increase in the value of the independent variable, while positive values indicate an increase in probability. Slope was for all land use types an important factor explaining the spatial location, in the case of crops a negative one, while for reforested areas and semi-natural vegetation a positive one. Another important factor for almond/olives was the presence of irrigation. Lithology was also a driving factor; almonds/olives, grapes and abandoned fields are mainly found on marls, while reforested areas were primarily found on Keuper. The standardized beta-coefficients show that for almonds/olives a low slope and the presence of irrigation were most important, while for forest low solar radiation, no flat position and further away from the main road were the most important factors explaining the spatial location. For abandoned land the standardized coefficients were less pronounced, which indicates that there was not one driving factor that determined the distribution, but a combination of several driving factors. The ROC-values for abandoned land and semi-natural vegetation were lower, which means that the driving factors do not explain the spatial pattern of these land use types completely. For the other land use types the ROC was rather high ranging from 0.83 till 0.96, which indicates a good fit of the logistic regression model.

Table 2.4. Driving factors and the standardized beta-coefficients for each land use type

Driving factor	**Almonds / olives**	**Cereals**	**Grapes**	**Abandoned**	**Semi-natural**	**Forest**	**Reforested**	**Built-up**
Altitude	0.078	0.156	-0.051		-0.161	0.093	-0.078	-0.044
Slope	-0.458	-0.554	-0.146	-0.149	0.318			0.026
Solar radiation				0.089	0.232	-0.282	0.068	
Flat position	0.060	0.071			-0.070	-0.349	-0.240	
Valley position			-0.049					
Slope position	0.158	0.109		0.022				0.006
Top position			-0.234		0.082			
Marl	0.057	0.255	0.043	0.017	-0.124			
Keuper						-0.088	0.067	
Limestone				-0.048				
Housing density	0.180	-0.088		0.024		-0.246	-0.134	
Distance to main road	0.063		-0.074		-0.186	-0.553	0.261	-0.005
Wetness index							-0.062	-0.010
Irrigated area	0.292						-0.367	
Distance to road	-0.052	-0.048		0.037	0.124			-0.063
Travel time to town		0.065	0.026	-0.019			-0.152	
ROC	*0.90*	*0.94*	*0.91*	*0.81*	*0.73*	*0.96*	*0.83*	*0.91*

Figure 2.3 shows the predicted land use maps for the four scenarios. In all scenarios the cereals only remain in the southwest corner of the study area on relatively flat marls that have no irrigation possibilities. The almonds and olives increase or remain mainly in the central flat parts that have irrigation possibilities and in the southeast of the study area. New abandoned land mainly occurs in the central southern part near the borders of the reforestation areas. The two scenarios with a large increase of abandonment (BB and CC-scenario) also show that small fields in the central part are abandoned and agriculture only remains on large flat fields.

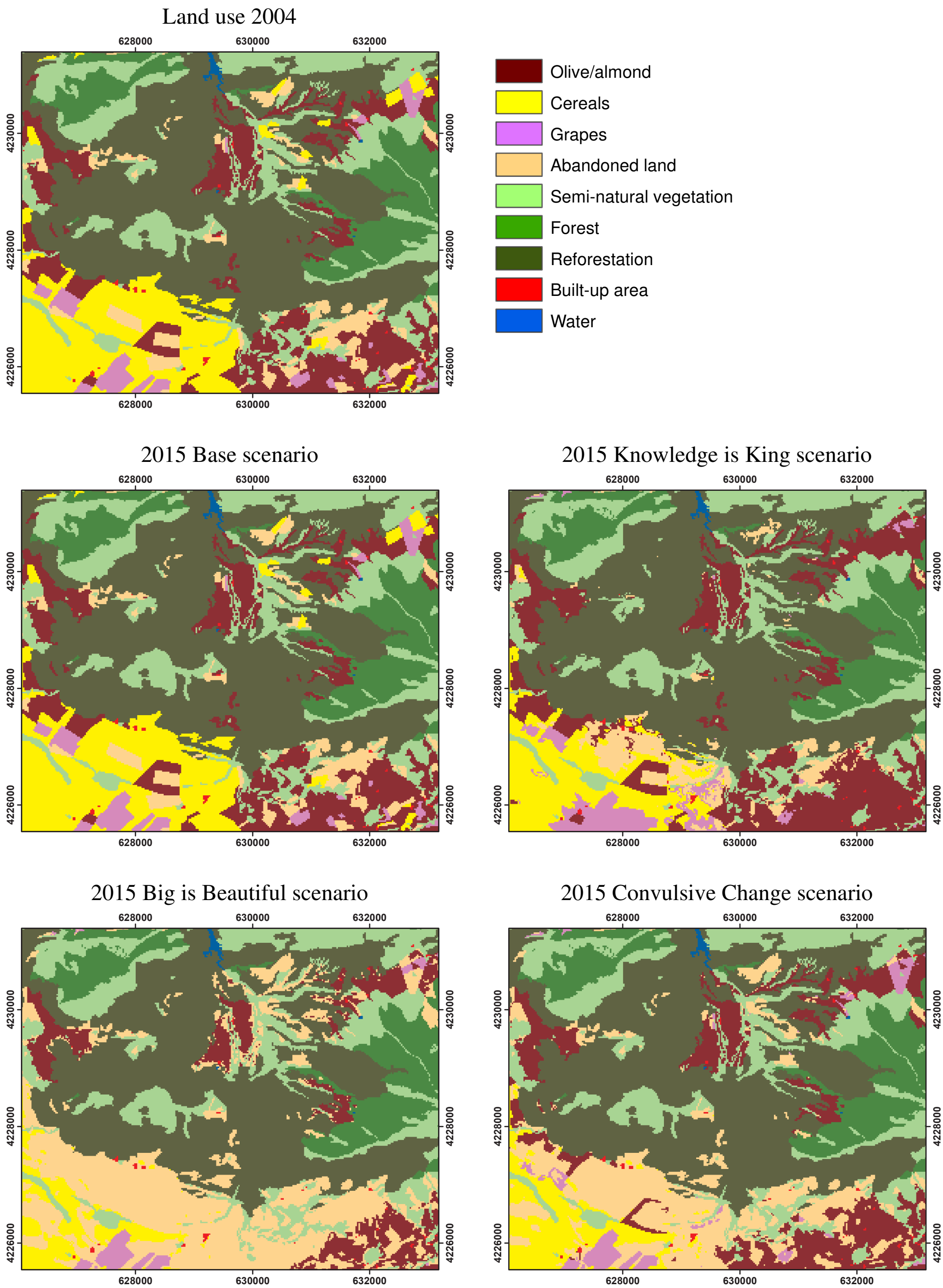


Figure 2.3. Current land use and simulated land use maps for 2015

2.4.3. Vulnerable areas for gully erosion

Based on the land use map of 2004 the area vulnerable to gully erosion consists of 61 ha out of 206 ha of abandoned land. When land abandonment is simulated the vulnerable area to gully erosion changes and an additional area of 26 ha for the base scenario, 18 ha for the KK-scenario, 176 ha for the BB-scenario and 78 ha for the CC-scenario become vulnerable to gully erosion (Figure 2.4). Most of the vulnerable areas are located around channel heads or along the channel walls.

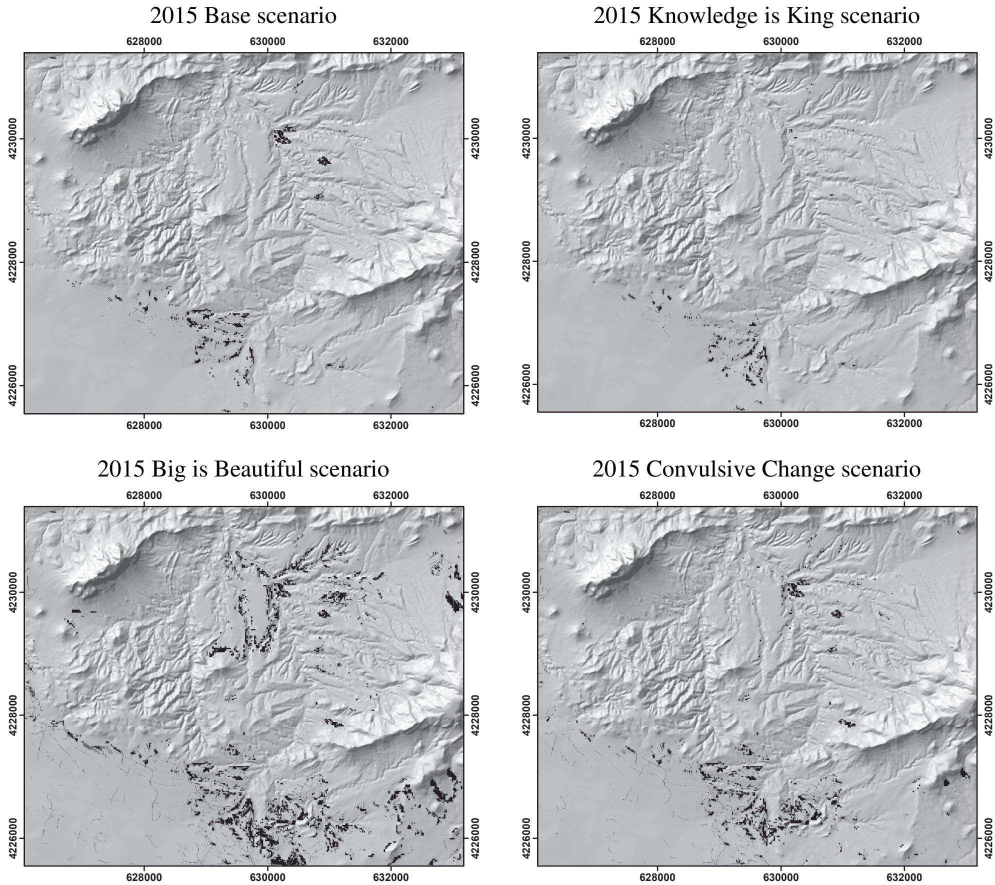


Figure 2.4. New vulnerable areas for gully erosion (in black) with a hill shade map as background

2.5. Discussion

2.5.1. Gully erosion

A comparison of gullies on abandoned and cultivated land is complicated, because cultivated land is under management, which might remove traces of a gully, i.e. by ploughing and filling of newly formed gullies. However, especially bank gullies, which often have a vertical headwall from the channel up to 5 meter depth, should remain clearly visible since they are difficult to fill. During the survey no remnants of recently filled gullies were found, only in some gullies pruned almond branches were found, which probably decreased the gully activity. Furthermore around some gullies and along some field boundaries small earth dams were constructed to prevent runoff to enter the gully. Oostwoud Wijdenes et al. (2000) also found these control dams and suggested that they were responsible for the poor statistical relationship between land use and gully activity. This supports the assumption that gully erosion is higher on abandoned fields, because well maintained soil conservation measures on cultivated fields reduce gully erosion.

Normally one would expect more gully erosion in tilled soil (Poesen et al., 2003), since the critical flow shear stress values are lower for loose soils (Franti et al., 1999). However, surface storage and infiltration capacity also improves in tilled soils, which increases the threshold for runoff occurrence. Since the soils in the area are very susceptible to slaking the farmers plough their almond and olive groves frequently, up to five times per year, to prevent slaking of the soil and suppress weed growth (van Wesemael et al., 2003). When abandoned fields are not ploughed anymore the soil surface starts to slake and a crust is formed. This has also been observed in sandy Sahelian soils where crusts developed during fallow periods as a result of dust deposition and colonisation by blue green algae (Valentin et al., 2004), which enhanced gully development. Similar crusting processes might also explain why widespread abandonment of communal cultivated fields in South Africa has been associated with gully initiation and intensification (Kakembo and Rowntree, 2003).

Other studies that indicate higher erosion risks at abandoned fields are Lasanta et al. (2000) who obtained a very quick response to precipitation, high peak flows, and a shallow wetting front on abandoned fields, which confirmed the effects of low density of plant cover and the development of a micro-crust. Oostwoud Wijdenes et al. (1999) found that the type of gully head changes from gradual on cultivated land to rill-abrupt and abrupt on abandoned land, which indicates an increase in sediment production from abandoned lands. Finally, Cammeraat et al. (2005) observed an increased landslide risk on abandoned slopes in a more humid part of Spain.

2.5.2. Land abandonment modelling

The ROC-values of the logistic regression models ranged from 0.73 till 0.96, which is fairly high for land use change modelling, given that Pontius and Schneider (2001) consider values above 0.7 as acceptable and values beyond 0.8 as excellent. Especially natural forest has a very high ROC-value, which indicates that its location can be very well predicted. On the contrary semi-natural vegetation and abandoned land are more difficult to predict, since these land use types can be considered as ‘remaining’ land uses, leading to lower ROC-values. Especially semi-natural vegetation can be found at diverse locations in the landscape, ranging from channel bottoms in marls to top positions on limestone mountains. This diversity makes it difficult to obtain driving factors that are valid for the whole area. A possible solution is the use of a predefined suitability variable as driving factor. Besides the drivers of the different agricultural land uses are diverse, and abandoned land originates from these different agricultural land use types, which makes it a mixed bunch of land, resulting in a lower ROC value.

The results of the multiple resolution procedure for the model goodness-of-fit after calibration (Figure 2.5) show that the simulation for 1985-2004 performed better than the null model. This means that the overall proportion of agreement between the actual land use map of 2004 and the simulated land use map is higher than for the null model. The simulation performed better for larger window sizes, which indicates that regional land use is simulated well, but at grid cell scale the simulation performed less. Local land use changes are more difficult to predict due to decisions of individual land users, while regional land use changes are easier to predict since these are more related to spatial driving factors. Pontius et al. (2008) compared nine different land use models and concluded that only half of the models were able to predict more accurately all land use changes than the null model at the fine resolution of the raw data.

The projected land use maps show a coherent visualization of possible land use distributions in 2015. The locations of newly abandoned areas are in agreement with the areas one would expect to be abandoned knowing the study area. Small fields without irrigation possibilities, located on slopes and valley bottoms, and on more remote places are abandoned first, which is also described in literature (MacDonald et al., 2000). Of all scenarios the CC-scenario is probably closest to the present-day reality, bearing in mind the expected increase of land abandonment due to climate change and changing European policies (Olesen and Bindi, 2002). However, unexpected changes and diversity in scenarios are important in order to cover the whole range of possibilities (Kok et al., 2006), therefore we also included the effects of a more extreme BB-scenario and the desired KK-scenario.

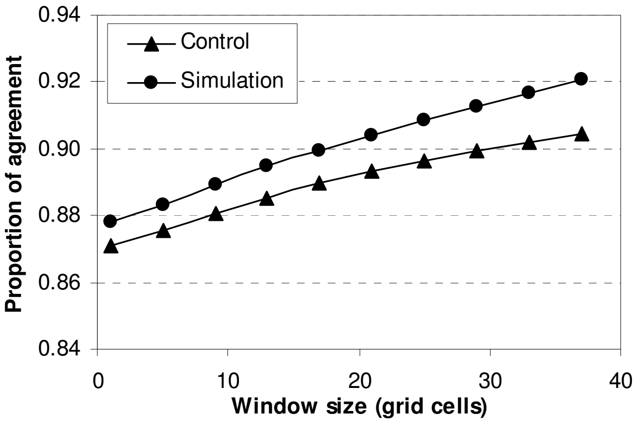


Figure 2.5. Plot of window size versus proportion of agreement

The prediction of land abandonment as presented in our paper illustrated that dynamic modelling of the interaction between spatial and temporal dynamics enables us to translate linear changes in land use requirements into non-linear impacts on the functioning of the landscape, e.g. gully erosion. A limitation of the model is that household/farm specific conditions are not incorporated, which can have significant consequences, e.g. once a farmer decides to retire his land might be abandoned, but the model will not allocate this change to the fields belonging to the farm, but to the location with the highest probability at regional level.

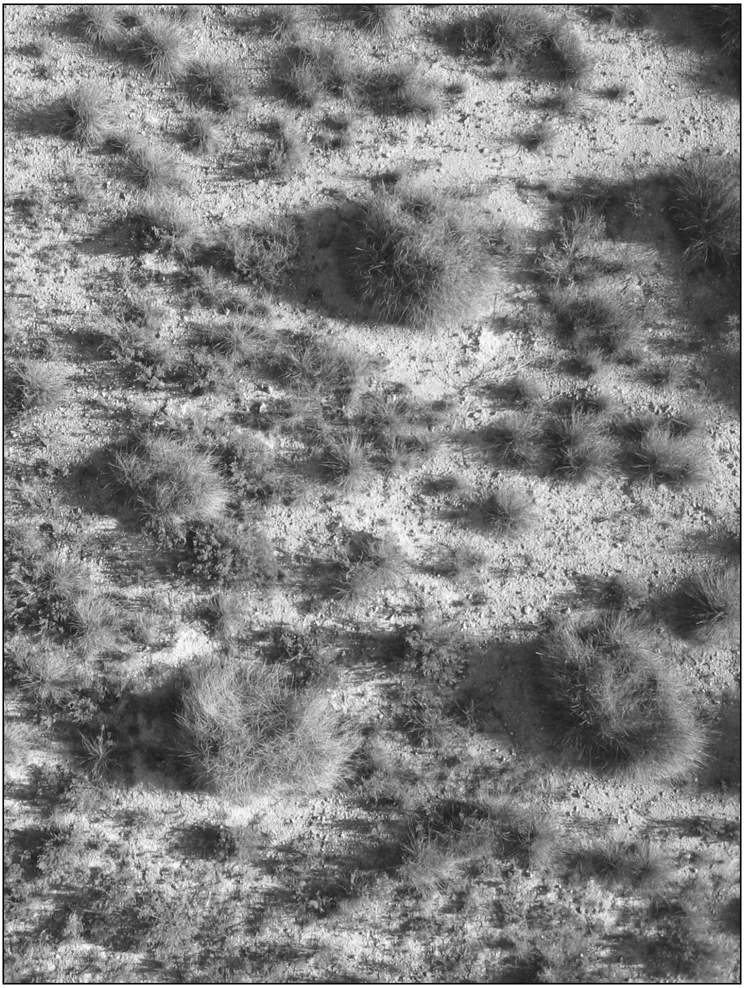
2.5.3. Vulnerable areas for gully erosion

Most of the areas vulnerable to gully erosion are located around channel heads or along channel walls, i.e. potential bank gullies, which is logical considering the steep gradient at these sites. Obviously the areas are potentially vulnerable and in reality gullies will only develop on a fraction of the surface. Nevertheless, only a small drainage area is necessary for the development of a gully due to the steep gradient along the channel (Vandekerckhove et al., 2000a). The exact location of soil and water conservation measures should be determined in the field where local topography determines the exact location of potential gullies. Furthermore, input data at higher resolution, especially the DEM, would improve the identification of vulnerable areas. Some areas, mainly in the central southern part, are vulnerable areas in all four scenarios. This is an indication that these areas are very likely to be abandoned. Especially there preventive soil and water conservation measures can be taken in advance.

We focused in our study only on areas vulnerable to gully erosion due to land abandonment. Since in cultivated fields already some type of soil conservation management occurs, e.g. regular ploughing and small ditches around fields, which mitigate gully erosion. On the contrary gullies in semi-natural areas already reached a certain steady state condition, and no major increase in erosion rate is expected. Whereas in abandoned land the process of gully erosion is just about to start, which means that potentially much more erodible sediment is available.

2.6. Conclusion

Gully erosion in semi-arid areas, like Southeast Spain, is higher on abandoned fields compared to cultivated fields. A slow vegetation recovery, soil crusting, and lack of maintenance of agricultural terraces explain this increase in gully erosion after abandonment. We project an increase of abandoned land, which will mainly occur on fields that have no irrigation possibilities, are located on slopes or in valley bottoms, and in more remote areas. However, these areas correspond with areas that have a higher gully erosion risk, since most of the vulnerable areas are located around channel heads or along channel walls. The combination of more gully erosion on abandoned fields and the expected increase of land abandonment is potentially a big problem in relation to land degradation and reservoir sedimentation. Nevertheless, the identification of potentially vulnerable areas enables soil conservationists and engineers to timely mitigate gully erosion by applying preventive soil and water conservation practices, e.g. revegetation of terrace edges. The visualisation of the spread of potential future land degradation may be an incentive for planners and policy makers to consider alternatives for abandoned agricultural areas.



3. Development of spatial heterogeneity in vegetation and soil properties after land abandonment in a semi-arid ecosystem [*]

3.1. Introduction

Agricultural land abandonment is one of the main changes in land use in Mediterranean countries, due to changing EU policies, urbanisation, globalisation and desertification (Geeson et al., 2002). A further increase of land abandonment is expected in the near future (Rounsevell et al., 2006). The cessation of extensive farming has led to substantial increase in dry grasslands and dwarf shrublands in marginal lands (Hernández, 1997). These abandoned fields can be vulnerable to erosion due to sparse initial vegetation cover, unfavourable soil properties and lack of maintenance of soil and water conservation structures (Gallart et al., 1994; Imeson et al., 1998). For mitigation of erosion, it is important to understand how vegetation and soil properties of these abandoned fields change and how vegetation patterns develop. Further, changes in vegetation patterns may be an indication of the onset of desertification in arid areas (Kéfi et al., 2007).

The secondary succession after abandonment initially starts with annual or biannual plants and is followed by perennial forbs, grasses and shrubs. Bonet (2004) found that annual plants and short-lived perennials have a higher cover and species richness during the first phase of abandonment, forbs during the second phase and finally woody species after 10 years of abandonment. However, forest may not be representative of later stages of succession under the semi-arid conditions of Southeast Spain, due to sparse water resources and intensive human disturbances in the landscape over millennia. A late successional community composed of shrublands is more likely for these areas (Rivas-Martínez, 1987). Nevertheless, succession is a highly variable process, which is influenced by many factors, such as environmental conditions, seedbank status, land use history, and plant population and community processes (Bonet, 2004; Flinn and Vellend, 2005). As a result of changes in vegetation and soil management, soil properties of abandoned fields will change as well after abandonment. In general a progressive recovery of vegetation cover, litter production, organic matter, water retention capacity and stability of aggregates takes place on

[*] Published as Lesschen, J.P., Cammeraat, L.H., Kooijman, A.M. and Van Wesemael, B. In press. Development of spatial heterogeneity in vegetation and soil properties after land abandonment in a semi-arid ecosystem. Journal of arid environments.

abandoned fields (Martinez-Fernandez et al., 1995; Kosmas et al., 2000; Dunjó et al., 2003; Bonet, 2004).

In semi-arid ecosystems, the lack of available water leads to sparse vegetation often in spotted or banded patterns (Tongway et al., 2001). The positive feedback between vegetation and water infiltration coupled with the spatial redistribution of runoff can explain the formation of these vegetation patterns (HilleRisLambers et al., 2001). As a result of this positive feedback, soil properties become spatially heterogeneous as well with more organic matter and improved soil physical properties under vegetation patches. This increased heterogeneity of soil resources is also known as the 'islands of fertility' phenomenon (Schlesinger et al., 1990). Hence, vegetation is the one of the key factors controlling overland flow generation, where runoff from bare patches infiltrates into vegetated patches (Ludwig et al., 2005), which makes overland flow highly discontinuous (Cerdà, 1998a; Lavee et al., 1998). Furthermore, the distribution of the vegetation patches determines whether runoff becomes connected at the hillslope scale, which will increase hydrological connectivity and the erosive force of runoff, but this also depends on the magnitude of the rainfall (Bracken et al., 2008).

Vegetation patterns in semi-arid grasslands and shrublands have been studied extensively (Bochet et al., 1999; Cammeraat and Imeson, 1999; Valentin et al., 1999; Maestre and Cortina, 2002). However, the spatio-temporal development of vegetation patterns is still largely unknown, and only simulated by models (Thiery et al., 1995; HilleRisLambers et al., 2001; Rietkerk et al., 2002). Alternatively, abandoned fields with different ages of abandonment offer the possibility to study the spatio-temporal development of vegetation patterns. Using an integrated approach, which combines secondary vegetation succession, changes in soil properties and analysis of vegetation patterns, the development of spatial heterogeneity can be described. Another issue that has to be addressed is the spatial scale of vegetation patterns. Although some vegetation patches can be large, e.g. groves of mulga trees (100–1000 m^2), such as observed in central Australia (Dunkerley, 2002), most vegetation patches in Mediterranean semi-arid shrublands are typically in the order of 1 m^2 or less. Given that the highest resolution of standard aerial photographs or satellite images is in the order of one metre, images with a high resolution may be indispensable for describing vegetation patterns correctly. This is particularly important as the spatial configuration of vegetation largely determines processes such as runoff and erosion, at least for events with a return period of up to 10 years (Bergkamp et al., 1996; Cammeraat, 2004; Ludwig et al., 2005).

The objective of our paper was to investigate the development of spatial heterogeneity in vegetation and soil properties after land abandonment in Southeast Spain. To achieve this objective, the following questions were answered. How does the vegetation composition evolve? How do topsoil properties change after land abandonment? How do vegetation patterns develop after land abandonment? The Carcavo basin in Southeast Spain was selected as study area, because of the presence of different successional stages of abandoned fields and the existence of distinct vegetation patterns as a result of the semi-arid climate. We selected two series of abandoned fields, one on marl and one on calcrete, which are the two main substrates used for agriculture within the catchment. For the selected field sites, we described the vegetation, collected soil samples and made detailed aerial photographs. These images were classified into bare and vegetated patches and several spatial metrics were calculated for each plot. Finally, we linked vegetation succession, change in soil properties and vegetation patterns to the development of spatial heterogeneity after land abandonment and discussed the implications for runoff and erosion.

3.2. Materials and methods

3.2.1. Study area

The Carcavo basin is located about 40 km northwest of the city of Murcia in Southeast Spain, near the town of Cieza (UTM coordinates 4228000 m N; 630000 m W; European_1950 datum zone 30N). It is a small catchment of 30 km^2 with altitudes ranging between 220 and 850 m. This region of Spain is very dry with an average annual rainfall of 300 mm and a potential evapotranspiration of 900 mm. The geology of the area consists of steep Jurassic carbonate mountains with calcareous piedmonts, Keuper gypsum marls and basin deposits of Cretaceous and Miocene marls. Most soils in the area are thin, weakly developed or characterised by their calcareous or gypsiferious parent material. According to the WRB classification, these soils are Leptosols, Regosols, Calcisols and Gypsisols (IUSS Working Group WRB, 2006). Current land use in the study area consists of agricultural land (barley, olives, almonds and vineyards), abandoned agricultural land, reforested land and natural land. In the 1970s, large parts were planted with pine (*Pinus halepensis*) for reforestation and soil conservation. During the last few decades, parts of the non-irrigated agriculture have been abandoned and are currently under different stages of secondary succession. The semi-natural vegetation on slopes is mainly composed of *Rhamno lycioidis-Quercetum cocciferae* shrubland in upper areas, while *Stipa tenacissima* and dwarf-shrub communities are dominant in lower areas.

3.2.2. Field sampling

Based on the space for time substitution approach (Paine, 1985) to overcome the problem of long term monitoring, we selected two sequences of abandoned fields (fallow, recently abandoned, long abandoned and semi-natural vegetation) in the south-eastern part of the Carcavo basin. One sequence of fields was located on a Cretaceous marl substrate (M sites) and the other sequence on Quaternary colluvial pediment with calcrete, which was partly broken by ploughing (C sites). The main difference between the two substrates was the amount of rock fragments on the soil, which influences crust formation and infiltration (Poesen and Lavee, 1994). All fields were used for traditional barley-fallow rotation before abandonment, indicating similar initial conditions. At each field, we selected a representative area of 10 × 10 m in the centre of the field on which all species were identified, using Sánchez Gómez and Guerra Montes (2003). Plant species were classified according to their growth form, i.e. annuals, herbs, grasses, shrubs or trees. At each plot, rock fragment cover (> 2 cm) was estimated, crust type was described according to Valentin and Bresson (1992) and slope and slope form were recorded (Table 3.1). The age of abandonment was estimated using aerial photographs of 1956, 1985, 1997, 1999 and 2002 and by asking the owner of the fields. None of the fields showed signs of disturbance by grazing, and according to the owner, the fields were not grazed since abandonment.

Table 3.1. Description of the field sites

ID	**Substrate**	**Stage**	**Rock fragment cover** (%)	**Crust type**	**Slope** (%)	**Slope form**	**Main vegetation species**
M1	Marl	Fallow	5	Slaking	4	Straight	Annuals
M2	Marl	± 6 yr abandoned	5	Slaking	0	Concave	*Bromus rubens*, *Eryngium campestre*, *Artemisia herba-alba*, *Plantago albicans*
M3	Marl	± 25 yr abandoned	14	Slaking	0	Concave	*Plantago albicans*, *Artemisia herba-alba*, *Lygeum spartum*
M4	Marl	Semi-natural (>50 yr)	13	Cryptogamic	3	Convex	*Quercus coccifera*, *Brachypodium retusum*
C1	Calcrete	Fallow	46	Slaking	5	Convex	Annuals
C2	Calcrete	± 9 yr abandoned	88	Sieving	1	Convex	*Helichrysum stoechas*, *Artemisia herba-alba*, *Teucrium capitatum*
C3	Calcrete	± 40 yr abandoned	35	Sieving and cryptogamic	2	Straight	*Stipa tenacissima*, *Thymus* sp., *Teucrium* sp., *Artemisia herba-alba*
C4	Calcrete	Semi-natural (>50 yr)	75	Sieving and cryptogamic	15	Straight	*Rosmarinus officinalis*, *Stipa tenacissima*

3.2.3. Soil sample analysis

On each field, four topsoil samples (0–5 cm) were taken from bare patches and four from vegetated patches. These soil samples were analysed for organic carbon, aggregate stability, pH and electrical conductivity. Furthermore, for each field one topsoil sample from a bare patch was analysed for $CaCO_3$ content, soil texture, micro-aggregation and sodium absorption ratio (SAR_p). For all these analyses, except for aggregate stability, the fraction <2 mm was used. Organic carbon content was determined with the wet oxidation method using $KCrO_6$ followed by colorimetric analysis (Page et al., 1982). Electrical conductivity (EC_{25}) and pH_{water} were measured in soil extracts of a 1:1 weight ratio. We determined aggregate stability of pre-wetted (pF 1) macro-aggregates (4–4.8 mm) with the water drop test (Imeson and Vis, 1984) for 20 replicates by counting the number of drops required to destroy the aggregate. $CaCO_3$ content was determined with the method of Wesemael (1955), which is based on weight loss on dissolution. Soil texture was analysed by wet sieving over a 106 μm sieve and measurement of the primary grain size distribution of the <106 μm fraction with a Sedigraph 5100, which determines grain size distributions by measuring settling velocity of particles with X-rays. In order to obtain the natural grain size distribution, the fine earth fractions were not decalcified, since $CaCO_3$ levels were above 50%. The percentage water-stable micro-aggregation was determined for the <106 μm fraction by subtracting the primary grain size distribution from the water stable grain size distribution (Edwards and Bremner, 1967; Cammeraat and Imeson, 1998). Concentrations of Na, Ca and Mg in a 1:1 soil-water extract were determined with an ICP-OES from which the practical sodium absorption ratio (SAR_p) was calculated (Sposito and Mattigod, 1977). Soils with SAR_p values above 2 are normally considered to be affected by salts and vulnerable to dispersion.

3.2.4. Statistical analysis

To analyse the variance of the soil property data we used a linear mixed models procedure, because one of the assumptions for analysis of variance with a general linear model was not satisfied. The replicates in our sampling design were collected within each plot and were therefore not completely independent. A linear mixed models procedure allows the data to exhibit correlated and non-constant variability and provides the flexibility of modelling not only the means of the data but their variance and covariance as well. The model parameters are estimated by minimisation of the weighted least-squares sum, and the parameters of the covariance matrix are estimated with a restricted maximum likelihood analysis (Webster and Payne, 2002). We used the linear mixed models procedure in SPSS 15.0 (SPSS Inc., Chicago, US) to test for significant factors for organic carbon, pH and EC. Substrate, stage and cover were set as fixed effects in the model and F tests were calculated on the basis of Wald statistics, with treatment effects declared significant at $P<0.05$. The aggregate stability data were non-normally distributed and therefore analysed with non-parametric tests (Siegel and Castellan, 1988). The Mann-Whitney test was used to compare bare and vegetated patches within each plot and to test for the overall influence of substrate. The Kruskal-Wallis test was used to compare the different stages of abandonment. Differences were declared significant at $P<0.05$.

3.2.5. Vegetation pattern analysis

Images with a high resolution were necessary to analyse vegetation patterns. Conventional aerial photographs have a resolution of 1 m, which is too low to distinguish the different vegetation patterns. Therefore, we used a balloon-mounted camera system (Ries and Marzolff, 2003) to make detailed aerial photographs of vegetation patterns on each site. Photos were made in April 2006 with a Canon Digital IXUS 500 camera attached to a balloon filled with helium. The camera was activated with a radio transmitter, which allowed us to take detailed aerial photographs from 30–50 m above the surface. These images had a spatial resolution of 1–2 cm, which made them suitable for the analysis of detailed vegetation patterns. For each site, we extracted a representative part of 10 × 10 m from the digital photo, which was resized to a pixel resolution of 2 cm. Next, the images were classified into bare and vegetated patches. Part of the bare patches on marl sites were covered with *Plantago albicans*. However, we considered these areas as bare patches, because rainfall simulations and soil analyses showed that sites with sparse *Plantago* cover were similar as bare patches, with low water infiltration and low organic carbon content. For the classification, we used the supervised maximum likelihood method of the remote sensing package ENVI (Research Systems, Boulder, US), which assigns each pixel to the class with the highest probability. This method classified the vegetation areas most accurately, as both the variance and covariance of the spectral response pattern are evaluated (Lillesand et al., 2004), which was confirmed by visual comparison with the original photo. In addition, this method was the only one that also classified the shaded areas correctly. Next, the classified images were filtered, using the ‘Median Filter’ with a filter size of 18 cm in Paint Shop Pro, in order to remove small areas of noise and to obtain a distinct pattern of bare and vegetated patches.

Afterwards, the images were analysed with FRAGSTATS 3.3 (McGarigal et al., 2002), a program to analyse spatial patterns of categorical maps. We calculated pattern metrics for the vegetation class using the default analysis and the eight-cell neighbourhood rule. The following metrics were calculated: percentage vegetation cover, mean patch size, patch density, largest patch index, edge density, area-weighted mean fractal dimension, connectance index, landscape division index and normalized landscape shape index (Table 3.2). The first five indices are default spatial metrics giving a measure of the fragmentation of the vegetation pattern (see e.g. Wu et al., 2002; Saura, 2004). The area-weighted mean fractal dimension indicates the complexity of vegetation patches using the same scale of measurement and assuming a self-similarity between patches of different size. Values close to 1 indicate patches with very simple shapes and small perimeters, while values approaching 2 indicate complex shapes with highly convoluted perimeters. The landscape division index is described by Jaeger (2000) as the probability that two randomly chosen places in the landscape are not situated in the same contiguous area. Advantage of this index is its insensitivity to omission or addition of very small patches. For our purpose, we calculated the index using the bare patches to indicate how strongly vegetation patches dissect the bare area, which is important for runoff generation (Imeson and Prinsen, 2004).

Table 3.2. List of spatial metrics as calculated by FRAGSTATS

Pattern metric	Description
Percentage cover	Ratio of the area of all vegetation to the total area (%)
Mean patch size	Average area of all patches (m^2)
Patch density	Number of patches per unit area (patch m^{-2})
Largest patch index	Ratio of the area of the largest patch to the total area (%)
Edge density	Total length of all edges per hectare (m m^{-2})
Area-weighted mean fractal dimension	Area-weighted mean fractal dimension of vegetation patches, which indicates the complexity of the patch shape: $AWMFD = \sum_{i=1}^{n}\left(\left(\frac{2\ln(0.25P_i)}{\ln(a_i)}\right)\left(\frac{a_i}{\sum_{i=1}^{n} a_i}\right)\right)$ where P_i and a_i are the perimeter and area of patch i and n is the total number of vegetation patches.
Landscape division index (calculated for bare area)	The index describes the probability that two randomly chosen pixels in an area are not situated in the same patch of the corresponding patch type: $LDI = 1 - \left(\sum_{i=1}^{n}\left(\frac{a_i}{A}\right)^2\right)$ where a_i is area of bare patch i, A is the total map area and n is the total number of bare patches.
Connectance index	Number of functional joinings between all patches divided by the total number of possible joinings between all patches (%): $CI = \left(\frac{\sum_{j \neq k}^{n} c_{ij}}{\frac{n(n-1)}{2}}\right) * 100$ where c_{ij} is the number of joinings between patch i and j and n is the number of vegetation patches.
Normalised landscape shape index	Normalised ratio between the total length of the edge and the minimum total length of edge: $NLSI = \frac{e_i - \min e_i}{\max e_i - \min e_i}$ where e_i is the total length of edge of class i in terms of number of cell surfaces.

Another related parameter is connectivity between vegetation patches, which influences transport of water and sediment through the landscape. In FRAGSTATS a proxy for connectivity is the connectance index, which is expressed as the percentage of the maximum number of possible joinings between vegetation patches. Each pair of patches is either connected or not, based on a user-specified distance threshold, which we set at 0.25 m. Finally, the normalised landscape shape index provides a measure of aggregation or clumpiness of vegetation patches. A value of zero means a maximally compact vegetation patch, while higher values indicate a more disaggregated vegetation pattern.

3.3. Results

3.3.1. Vegetation succession

Figure 3.1 shows the number and type of plant species which were found on each site. For the calcrete sites, a clear decrease in number of species with time since abandonment was found while for the marl sites, this decrease was only observed for the semi-natural site. The decrease in the number of species is mainly associated with the decrease in annuals and herbs. Although the number of shrub species hardly increased with time since abandonment, the shrub cover increased considerably. Fallow sites were mainly covered by annual herbs such as *Anagallis arvensis*, *Moricandia arvensis*, *Centaurea aspera* and *Filago pyramidata*, but also some small shrubs such as *Artemisia herba-alba* and *Teucrium capitatum* were found, but none of the species was dominant. On the recently and long abandoned sites on marl, patches of herbs (*Carrichtera annua*, *Eryngium campestre* and *Eruca vesicaria*, grasses (*Bromus rubens*, *Lygeum spartum* and *Dactylus glomerata*) and few shrubs (*Artemisia herba-alba* and *Salsola genistoides*) occurred with *Plantago albicans* on bare patches. The recently abandoned site on calcrete was dominated by *Artemisia herba-alba*, *Helichrysum stoechas* and *Teucrium capitatum*. The long abandoned site on calcrete had similar plant species as the semi-natural vegetation, such as *Stipa tenacissima* and *Rosmarinus officinalis*, but the occurrence of *Artemisia herba-alba* and some *Plantago albicans* indicated its former agricultural use. Finally, on the semi-natural sites, mainly *Stipa tenecissima* and *Rosmarinus officinalis* were found with some *Quercus coccifera* and *Pinus halepensis* and for the site on marl, also quite some *Brachypodium retusum* grass.

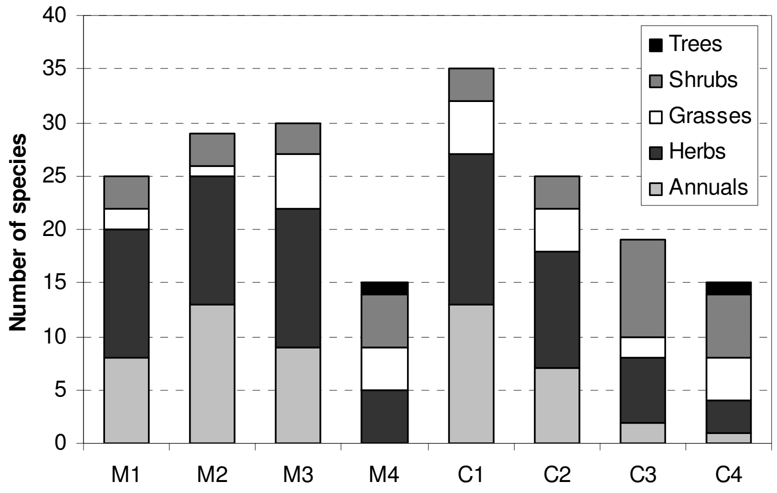


Figure 3.1. Number and type of plant species of the field sites

3.3.2. Soil properties

Table 3.3. Topsoil properties of the field sites under bare soil

ID	Substrate	Stage	$CaCO_3$ (%)	Org. C (%)	Sand (%)	Silt (%)	Clay (%)	SAR_p $(mmol/l)^{1/2}$	Micro-agg.[a]
M1	Marl	Fallow	60	0.78	30	56	14	0.13	21.0
M2	Marl	± 6 years abandoned	50	1.04	6	72	22	0.12	25.8
M3	Marl	± 25 years abandoned	49	0.84	16	64	20	0.10	17.9
M4	Marl	Semi-natural	50	1.14	23	64	13	0.11	13.7
C1	Calcrete	Fallow	67	0.99	32	56	12	0.11	19.4
C2	Calcrete	± 9 years abandoned	65	1.38	61	33	6	0.09	19.5
C3	Calcrete	± 40 years abandoned	69	1.02	20	63	17	0.11	26.0
C4	Calcrete	Semi-natural	56	2.12	50	41	9	0.08	17.2

[a] Micro-aggregation as percentage of the <106 µm fraction

All sites had a high $CaCO_3$ content, high silt content, low SAR_p value and similar micro-aggregation (Table 3.3). The main difference was the coarser texture on calcrete sites. Generally, organic carbon increased with time since abandonment, especially for vegetation patches (Figure 3.2). Obviously, vegetated patches had higher organic carbon content than bare patches. Aggregate stability showed a similar response with higher aggregate stability for vegetated patches (Figure 3.3). These differences between bare and vegetated patches were significant, except for site M2. Aggregate stability also increased with time since abandonment and the Kruskal-Wallis test showed that the differences between at least one of the stages was significant.

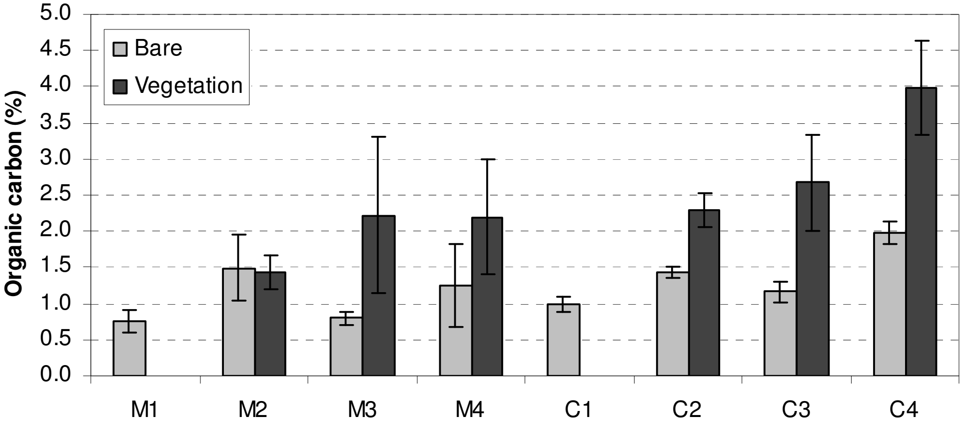


Figure 3.2. Organic carbon content for bare and vegetated patches (error bars indicate standard deviation, n=4)

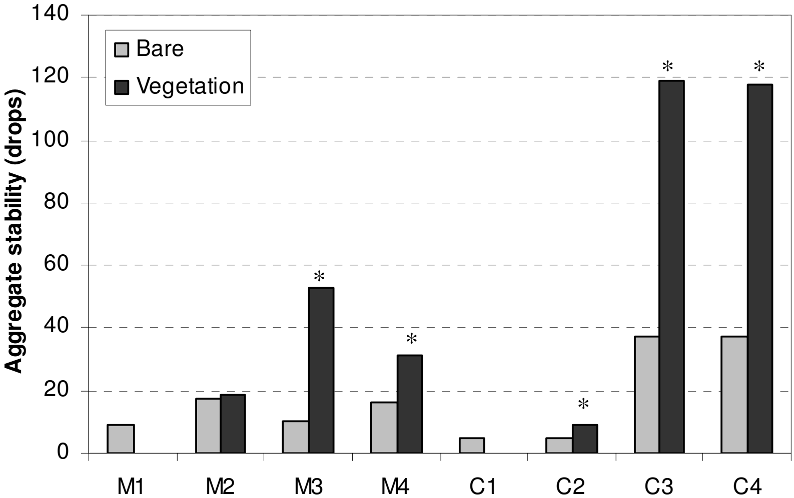
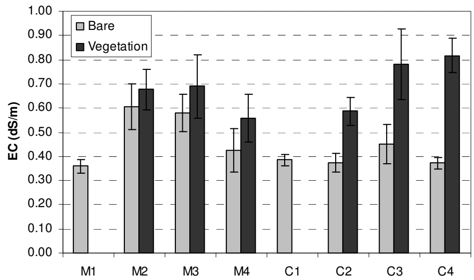


Figure 3.3. Aggregate stability for bare and vegetated patches (* indicates significant difference between bare and vegetated patches)

Electrical conductivity was lower under bare patches, with values around 0.40 dS m^{-1}, and higher under vegetated patches with values up to 0.82 dS m^{-1} (Figure 3.4). However, no clear trend related to time since abandonment was observed. Bare patches had approximately EC values around 0.40 dS m^{-1}, while for vegetated patches, an increase in EC with time since abandonment was observed for calcrete, but not for marl. Differences in pH were less pronounced, with values ranging from 7.0 to 7.4. No clear trend with time since abandonment was observed, but at each site pH was lower under vegetation. Table 3.4 shows the results of the linear mixed model procedure. Substrate had only a significant effect on organic carbon, while stage and cover had significant effects on all variables. The combinations of two factors also showed significant effects, except the combination stage and cover for EC, but the combination of all three factors was not significant for any of the three soil properties.



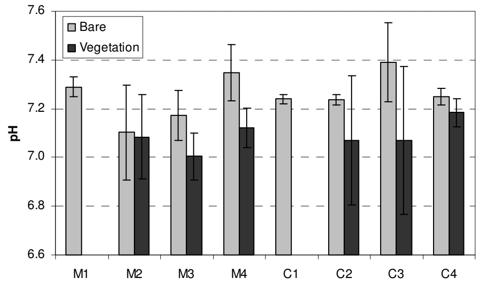


Figure 3.4. EC and pH for bare and vegetated patches (error bars indicate standard deviation, n=4)

Table 3.4. Tests of the fixed effects for organic C, pH and EC

Factor	**Organic C**		**pH**		**EC**	
	F-value	*P*-value	*F*-value	*P*-value	*F*-value	*P*-value
Substrate	67.69	0.000	1.03	0.323	0.35	0.577
Stage	19.84	0.003	9.93	0.000	8.34	0.012
Cover	78.73	0.000	56.70	0.000	95.24	0.000
Substrate × stage	9.27	0.015	5.46	0.007	5.79	0.030
Substrate × cover	35.36	0.002	6.47	0.020	7.94	0.028
Stage × cover	9.87	0.016	7.27	0.005	3.43	0.096
Substrate × stage × cover	2.85	0.143	1.53	0.228	0.85	0.472

3.3.3. Vegetation patterns

Figures 3.5 and 3.6 show the aerial photographs and derived vegetation patterns for the marl and calcrete sites. Percentage vegetation cover increased with time since abandonment and the vegetation patches became larger and more connected. The long abandoned site on marl still had a sparse vegetation cover, with relatively small patches, while the long abandoned site on calcrete already appeared similar to semi-natural vegetation. The spatial metrics showed for both substrates an increase of mean patch size, largest patch index, landscape division index and connectance index and a decrease of normalised landscape shape index with time since abandonment (Table 3.5). For the area-weighted mean fractal dimension no clear trend was observed. Patch and edge density were highest for the sites with the highest number of vegetation patches, which were the recently abandoned sites.

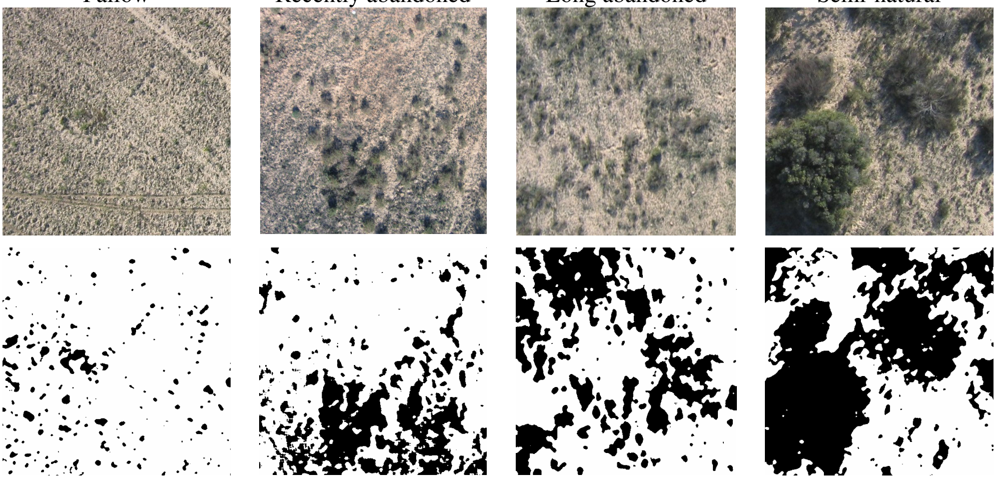


Figure 3.5. Vegetation patterns and cover for the marl sites

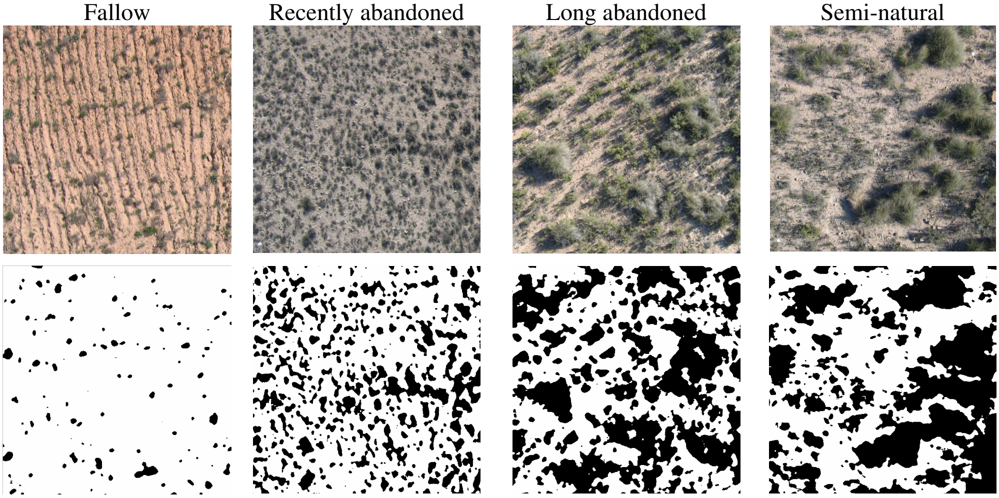


Figure 3.6. Vegetation patterns and cover for the calcrete sites

Table 3.5. Calculated spatial metrics

Description	M1	M2	M3	M4	C1	C2	C3	C4
Percentage vegetation cover (%)	6.1	25.5	32.5	51.5	4.1	27.4	45.9	46.0
Mean patch size (m^2)	0.03	0.15	0.32	0.90	0.03	0.09	0.42	0.42
Patch density (patch m^{-2})	1.74	1.78	1.01	0.57	1.32	2.94	1.09	1.09
Largest patch index (%)	0.87	9.80	7.81	44.8	0.32	2.33	13.4	22.2
Edge density (m m^{-2})	1.4	2.9	2.6	2.0	0.9	4.5	3.5	2.8
Area-weighted mean fractal dimension	1.11	1.25	1.19	1.24	1.07	1.16	1.22	1.20
Landscape division index (fraction)	0.12	0.47	0.57	0.89	0.07	0.48	0.80	0.73
Connectance index (%)	0.32	1.02	1.47	1.92	0.42	0.69	1.61	1.47
Normalized landscape shape index	0.11	0.06	0.04	0.02	0.11	0.08	0.04	0.03

3.4. Discussion and conclusions

Vegetation recovery after land abandonment in a semi-arid environment appears to be slow and to take at least 40 years, as indicated by the longest abandoned field (C3) which still had some plant species that are typical of abandoned fields such as *Artemisia herba-alba*. This recovery rate is much slower than the one in more humid environments in the Mediterranean, e.g. 9–15 years for an abandoned olive grove in Italy (Beaufoy, 2001), 8–15 years for an East Mediterranean vineyard (Neeman and Izhaki, 1996) and 10 years to reach a 100% vegetation cover on abandoned fields in the Spanish Pyrenees (Lasanta et al., 2006). Succession on marl appears to be slower than on calcrete, as observed by the relatively high shrub cover on the short abandoned site on calcrete (C2) and the degraded state of the long abandoned site on marl (M3). On this site the vegetation cover was relatively sparse and the number of annuals and total number of plant species is still high, which is an indication of the coexistence of many early successional species and only few late successional species (Bonet, 2004). However, due to differences in time since abandonment between the two substrates, we cannot state with absolute certainty that succession on calcrete is faster. The main reason for the difference between marl and calcrete is probably the higher rock fragment cover on calcrete sites, which allows for better infiltration, less runoff and lower evaporation rates (Poesen and Lavee, 1994; Van Wesemael et al., 1996; Cerdà, 2001). Consequently, more water is available for vegetation and a faster succession is possible.

During the first years of abandonment, vegetation is dominated by herbs, but later shrub and grass species such as *Rosmarinus officinalis* and *Stipa tenacissima* become more important. Surprisingly, the number of shrub species did not increase much with successional time. However, further examination showed that the shrub species on fallow and recently abandoned fields are mainly dwarf and half-shrubs, such as *Teucrium* *capitatum* and *Dittrichia viscosa*, which did not occur during the later stages of succession. For a more detailed description of vegetation succession, more plots are necessary, because occurrence of plant species is also determined by other factors such as composition of the soil seed bank, disturbance by grazing, micro-topography, crusting and type of vegetation surrounding the field (Pugnaire et al., 2006). Vegetation type might be particularly important, since most of the abandoned fields were small in size and close to some remnant parts of semi-natural vegetation, which can be an important source of seeds.

Our results confirm that the analysed soil properties after abandonment are able to recover to their level from before cultivation, which was also found by Martinez-Fernandez et al. (1995) and Ruecker et al. (1998). However, this recovery is slow and even after 40 years (site C3) soil organic carbon content is still lower compared with the semi-natural vegetation (site C4). Aggregate stability recovered more quickly and the long abandoned sites had aggregate stabilities comparable with the semi-natural sites. Cammeraat and Imeson (1998) and Cerdà (1998b) also found that soil aggregation increased with time since abandonment. Aggregates under late successional species were more stable compared with those under dwarf shrubs and bare soil.

Both substrates had similar soil properties, with comparable $CaCO_3$, SAR_p and micro-aggregation values, while only rock fragment cover was higher and soil texture somewhat coarser for calcrete. Differences in organic carbon content, aggregate stability, EC and pH could therefore be primarily attributed to the influence of vegetation. In general, vegetation succession was faster on calcrete compared with marl, which was also reflected in higher soil organic carbon content, aggregate stability and electrical conductivity of vegetation patches on calcrete. An important aspect is the difference in soil properties between bare and vegetated patches, which is typical of semi-arid ecosystems (Rietkerk et al., 2002), but not always taken into account. The difference in soil properties between bare and vegetated patches increases with time since abandonment, because no litter input occurs on bare patches, which impedes the accumulation of soil organic carbon. On vegetated patches, however, a clear increase of organic carbon and aggregate stability occurs. This also has consequences for sampling designs and applicability of soil property results. For example, for a correct estimation of organic carbon stocks in the topsoil, a weighted average of samples under bare and vegetated patches should be used.

The analysis of vegetation patterns showed that percentage vegetation cover, size of vegetation patches and connection between patches increase with time since abandonment. This development can be explained by vegetation succession, where small patches of herbs and dwarf shrubs are replaced by larger patches of grasses and shrubs, such as *Stipa*, *Rosemarinus* and *Quercus coccifera*. Thus, the spatial heterogeneity of the soil increases due to the positive feedback between vegetation and water infiltration coupled with the spatial redistribution of runoff (HilleRisLambers et al., 2001). Shoshany and Kelman (2006) describe the different evolution stages of pattern formation using modelled vegetation maps, based on cellular automata, which are in agreement with our results.

We also found a very clear relationship ($P<0.05$) between percentage vegetation cover and most spatial metrics, except patch and edge density (Table 3.6). Such a strong relationship was also found by Imeson and Prinsen (2004) for vegetation patterns of *Stipa tenacissima* shrublands. This means that percentage vegetation cover is a very good proxy for many spatial metrics of spotted vegetation patterns. Quinton et al. (1997) found that from all considered plant properties, only percentage vegetation cover had a significant relationship with runoff and erosion. Similarly Gutierrez and Hernandez (1996), Cerdà (1998a) and Descheemaeker et al. (2006) emphasise that vegetation cover is a key factor controlling overland flow generation, at least for events with a return period of up to 10 years. This implies that in many of these patchy vegetation landscapes, percentage vegetation cover is a good indicator of the risk of runoff generation and consequently erosion, though for banded vegetation, this might be different. The effect of patchy vegetation on runoff production is especially relevant for smaller and intermediate rainfall events, whereas the effect is smaller for the large rainstorms (Cammeraat, 2004). Several studies found that a threshold of 30% vegetation cover is already sufficient to decrease soil erosion considerably (Francis and Thornes, 1990; Rogers and Schumm, 1991; Quinton et al., 1997). Ludwig et al. (2002) showed that below 30% vegetation cover, the 'directional leakiness index', i.e. potential runoff, increased sharply. For abandoned fields in the Carcavo basin this would mean that after 10–20 years, vegetation cover would be sufficient to decrease soil erosion.

Table 3.6. Pearson correlation between vegetation cover and spatial metrics

Spatial metric	Pearson correlation	Significance
Mean patch size	0.86	0.007
Patch density	-0.45	0.268
Largest patch index	0.79	0.019
Edge density	0.49	0.214
Area-weighted mean fractal dimension	0.82	0.012
Landscape division index	1.00	0.000
Connectance index	0.95	0.000
Normalised landscape shape index	-0.96	0.000

The implications of patchy vegetation distribution are illustrated by Cerdà (1997). His experiments showed that infiltration rates are high and runoff and erosion are negligible in the tussock of *Stipa tenacissima*, while on bare areas, infiltration was lower and erosion and runoff quite high. This allows for a division into bare runoff (source area) and vegetated runon (sink area) zones. This vegetation-driven spatial heterogeneity of sources and sinks of water and sediments is also well described by Puigdefabregas et al. (1999) and Puigdefabregas (2005). At hillslope scales the extent of connectivity between these source areas determines whether runoff will occur. Puigdefabregas et al. (1998) showed that runoff coefficients may be locally high, but at the hillslope scale, they are low due to the spatial discontinuity of runoff with vegetation patches acting as sinks. Nevertheless, runoff, sediment yield and vegetation growth are also linked at the hillslope scale, and Ludwig et al. (2005) found that a decrease in vegetation cover leads to an increase in runoff and sediment loss. Lavee et al. (1998) showed that climate change and grazing can lead to changes in vegetation pattern, which affect the hydrological processes.

To mitigate runoff and erosion from abandoned fields, but also from semi-natural areas, it is important to account for differences in spatial heterogeneity of vegetation and soil properties. With our integrated approach, which combined the secondary vegetation succession, changes in soil properties and development of vegetation patterns, we now better understand the development of spatial heterogeneity in vegetation and soil properties for the sites under study. Upscaling of spatial heterogeneity is required to predict where runoff and erosion will occur at hillslope and catchment scales. The strong relationship that we found between vegetation cover and most spatial metrics makes it possible to upscale spotted vegetation patterns to larger areas. Together with the knowledge of the development of spatial heterogeneity in vegetation and soil properties, we can improve the predictions of runoff and erosion on abandoned fields and semi-natural areas. However, to be more confident with this approach a greater replication of sites is required.

4. Erosion and terrace failure due to agricultural land abandonment in a semi-arid environment *

4.1. Introduction

Abandonment of agricultural land is nowadays widely spread in Spain (Geeson et al., 2002; MacDonald et al., 2000; Fernandez-Ales et al., 1992) and most land use change scenarios project a further increase of land abandonment for southern Europe (Olesen and Bindi, 2002; Rounsevell et al., 2006). This conversion of cultivated land into abandoned land has also consequences for soil quality and erosion and sedimentation processes. Secondary vegetation succession after abandonment increases vegetation cover and improves soil properties, which will decrease runoff and erosion. However, in semi arid areas the recovery of semi-natural vegetation is limited by water and seed dispersal (Pugnaire et al., 2006), which results in a low vegetation cover. Large bare areas in combination with torrential storms enhance the development of soil crusts that reduce the infiltration capacity, which on its turn increases runoff and erosion.

While a decrease in erosion after land abandonment is observed in more humid parts of Spain (García-Ruiz et al., 1996; Molinillo et al., 1997), the semi-arid areas, e.g. Southeast Spain, show an increase in erosion during the first years/decades after abandonment (Cerda, 1997; Bull et al., 2000; Lasanta et al., 2000). Especially abandoned fields on marls are very vulnerable to erosion due to the erodibility of marls, which is caused by the dispersion of clay minerals, high swelling pressures and slaking (Bryan and Yair, 1982), and retarded germination of plant seeds (Garcia-Fayos et al., 2000). In addition, soil and water conservation structures, such as terraces, might collapse due to lack of maintenance and consequently increase erosion (Gallart et al., 1994). These agricultural terraces were built to retain more water and soil and to reduce the hydrological connectivity and erosion (Lasanta et al., 2001; Cammeraat, 2004). However, when the terraces are no longer maintained or when the water level behind the terrace wall passes the critical threshold during rain storms, terrace failure will occur and gully formation can start.

* Published as Lesschen, J.P., Cammeraat, L.H. and Nieman, T. In press. Erosion and terrace failure due to agricultural land abandonment in a semi-arid environment. Earth Surface Processes and Landforms.

Lesschen et al. (2007) found that in Southeast Spain especially small fields in valley bottoms that have no irrigation possibilities are prone to land abandonment. These fields are often terraced, which make them more prone to gully erosion. In addition, in case of dispersive substrate, the terraces are vulnerable to piping due to the presence of a steep gradient and horizontal impeding layers (Faulkner et al., 2003; Romero Díaz et al., 2007). According to Poesen et al. (2003) gully erosion is the main erosion process in terms of sediment production at the catchment scale. Koulouri and Giourga (2007) found that soil erosion increased significantly in terraced olive groves after abandonment, due to collapsing of the drystone terraces and lower protective vegetation cover. This was also observed by Cammeraat et al. (2005) for a more humid environment, where increased mass wasting processes after abandonment destroyed the bench terraces. Cammeraat (2004) found that soil erosion on terraced valley bottoms, based on an event basis, was about a hundred times higher compared to semi-natural hillslopes. For those reasons we focused our study on abandoned fields and especially fields with terraces, since these appear to have the highest erosion rates.

The objective of our study was to assess the extent and causes of erosion and terrace failure on abandoned fields and to discuss options for mitigation. To reach this objective we formulated the following questions. (1) What are the causes and extent of erosion on abandoned fields? (2) Which factors induce terrace failure? (3) What are possible soil and water conservation options to mitigate erosion on abandoned terrace fields? The Carcavo basin in Southeast Spain was selected as our study area, since it is representative for marginal agriculture with land abandonment in a semi-arid environment.

4.2. Study area

The Carcavo basin is located about 40 km northwest of the city of Murcia in Southeast Spain, near the town of Cieza (UTM coordinates 4228000 m N; 630000 m W; European_1950 datum zone 30N). It is a small catchment of 30 km^2 and altitudes range between 220 and 850 meter (Figure 4.1). This region of Spain is very dry with an average annual rainfall of 300 mm and a potential evapotranspiration of 900 mm. The geology of the area consists of steep Jurassic limestone and dolomite mountains with calcareous piedmonts, basins with Cretaceous and Miocene marls, and variegated Keuper deposits including gypsum. Most soils in the area are thin (Leptosols), weakly developed (Regosols) and mainly characterised by their parent material (Calcisols and Gypsisols). Land use comprises barley, olives, almonds, vineyards, abandoned land, reforested land and semi-natural vegetation. In the 1970s large parts of degraded land were reforested with pine (*Pinus halepensis* Mill.) within the framework of reforestation and soil conservation programs. During the last decades, parts of the non-irrigated agriculture have been abandoned and are under different stages of secondary succession. Abandoned land covers now 5 percent of the catchment, but accounts for 17 percent of the total agricultural land.

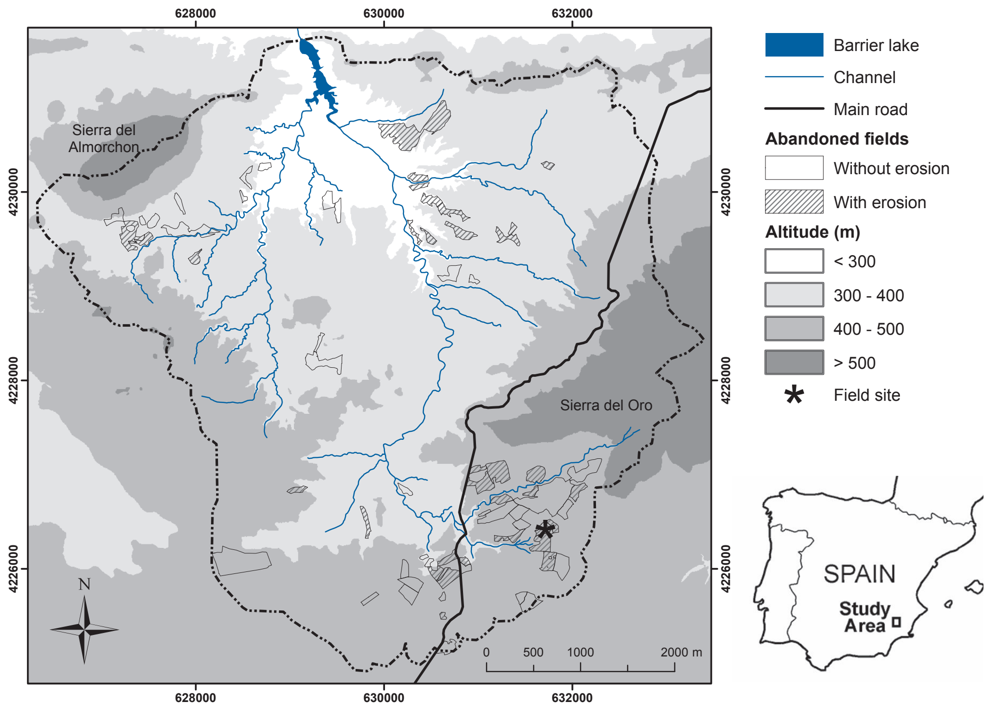


Figure 4.1. Study area of the Carcavo basin and the distribution of the abandoned fields

In the Carcavo basin about 39 percent of the agricultural land is currently terraced or has earth dams, which highlights the importance of this agricultural system in sloping areas. Terraces have been constructed on hillslopes and in streambeds, while earth dams are mainly found in valley bottoms of undulating cereal fields. Most of the terraces in the study area are level bench terraces as illustrated in Figure 4.2. The original aim of terracing is to reduce soil erosion and to intercept runoff by decreasing the general slope (Morgan, 1995). However, the effects of terracing are not only positive. Figure 4.2 shows the change of a slope profile after terracing, where area A represents topsoil material that is removed, which brings non-weathered material to the surface. This subsoil is often less fertile and more susceptible to erosion due to lack of organic matter and soil structure. For area B the situation is also ambiguous. Although the general slope angle is reduced, the area near the terrace rim is now under influence of a steep change in topography, where the hydraulic gradient of surface and sub-surface water may lead to gully erosion and piping (Faulkner et al., 2003; Romero Díaz et al., 2007).

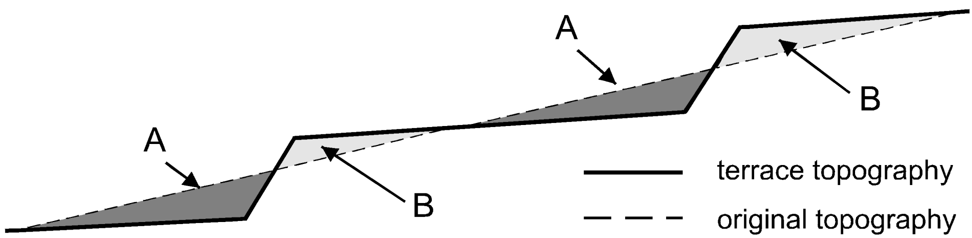


Figure 4.2. Artificial terracing of sloping terrain, (A) removed material and (B) accumulated material from upslope areas

4.3 Methodology

The field survey consisted of two parts. First we identified and described all abandoned fields in the study area. Second we surveyed both abandoned and cultivated terrace fields and described the terraces and the terrace failures. From aerial photographs and field surveys we identified in total 58 abandoned fields in the Carcavo basin (Figure 4.1). These fields were surveyed and the following properties were described: previous land use, parent material, vegetation, age of abandonment, erosion features, and presence of terraces or earth dams. With the GIS-software ArcGIS 9.0 (ESRI, Redlands, US) the field boundaries were digitised and by overlaying the map with a 5 meter resolution DEM the following properties were determined as well: surface, altitude, slope, potential drainage area and solar radiation. The 58 fields were then classified into abandoned fields with erosion and without erosion. To be classified as abandoned fields with erosion, the field should show signs of at least moderate erosion, being visible as gully, terrace failure or well developed rill. For gullies we used the definition of Poesen (1993), which defines gullies as rills with a cross-section larger than 929 cm^2. With SPSS 15.0 (SPSS Inc., Chicago, US) we tested whether the differences of the properties for the two groups were significant. The Pearson's chi-square test was used for binary variables and the t-test for continuous variables.

The analysis of terrace failure is based on a data set of 288 terraces within the Carcavo basin, both on cultivated and abandoned fields. From these terraces 121 were collapsed at the time of the survey in spring 2006. The following properties were described for each terrace: land use, geology, slope, topographic position, vegetation cover, rock fragment cover, texture, infiltration, estimated time since ploughing, terrace wall dimension, vegetation cover on the terrace wall and maintenance of the terrace wall. Texture and infiltration were both estimated in the field based on the guidelines for soil description (FAO-ISRIC, 1990) and the degree of soil crusting (Valentin and Bresson, 1992). Besides, electrical conductivity (EC) and aggregate stability of the soil were determined for every fourth terrace in a field. We determined aggregate stability of pre-wetted (pF 1) macro-aggregates (4-4.8 mm) with the water drop test (Imeson and Vis, 1984). Additionally, we recorded the coordinates of the centre of each terrace and terrace failure with a handheld Trimble GeoXM GPS (±2 m precision). Using GIS we derived the following properties for each terrace: potential drainage area, altitude and solar radiation. For altitude and solar radiation the mean value of a rectangular neighbourhood of 25 by 25 meter was used, and for potential drainage area the maximum value of the flow accumulation map of a rectangular neighbourhood of 45 by 45 meter was used.

For the analysis of the data we first compared the intact terraces with the failed terraces to identify which explanatory variables were significantly different, using the t-test for continuous variables and the Pearson's chi square test for binary variables. Afterwards, we created two logistic regression models for terrace failure prediction. For the first regression we used all variables and for the second regression we used all spatial variables, which allowed for a spatially explicit prediction of terrace failure. The logistic regression model was constructed using both forward and backward stepwise logistic regression. For a consistent regression model both methods should result in the same regression model. The resulting logistic regression coefficients were standardized according to Menard (2001), which allows for the comparison of the strength of the relationship between the dependent variable and different independent variables. To evaluate the performance of the regression model we used the Hosmer-Lemeshow test and the Relative Operating Characteristic (ROC). The Hosmer-Lemeshow goodness-of-fit statistic groups cases into deciles of risk and compares the observed probability with the expected probability within each decile. The ROC is a common measure for the goodness-of-fit of a logistic regression model, for an ideal model the outcome is 1, while a random variable achieves approximately 0.5 (Swets, 1988).

To determine the impact of land abandonment on soil erosion, the quantification of soil loss rates is essential in order to assess the severity and to target mitigation possibilities. A long monitoring period is necessary to obtain realistic long term soil loss rates, since most erosion occurs during the low-frequency high-intensity events (Cooke et al., 1993). An advantage of abandoned terrace fields is that most of the sediment is lost by gully erosion through the terrace walls, while sheet erosion is less important because of the low gradient on the terraces, which makes it simpler to reconstruct erosion rates. By determining the volume of lost sediment from the terrace failures and assuming that gully formation started after abandonment, the erosion rate can be calculated. As in the study area most of the gullies on terraces are from the abrupt gully head type (Oostwoud Wijdenes et al., 1999), the amount of lost sediment can be estimated relatively accurately.

For the detailed analysis of erosion rates we selected an abandoned field, which was located in a valley bottom just before a channel incision (Figure 4.1). This small field (0.6 ha) with five terrace levels on Cretaceous marl, was used for cereals and abandoned around 1984. The vegetation cover on the field was with 30-40% still low, which has led to crusting on the bare areas. The lack of maintenance, the increase in overland flow and the occurrence of piping led to terrace failure and gully erosion at 10 locations in the field. We surveyed the field site with a Trimble differential GPS system (±2 cm precision) to construct a detailed DEM. On average a survey point was recorded each 5 meter, but around the terrace walls and terrace failures the sampling density was much higher. With the 'Topo to Raster' procedure in ArcGIS, based on an algorithm of Hutchinson (1989), we constructed a hydrologically correct topography with a resolution of 0.5 meter. The interpolation was based on 1860 points and utilised drainage enforcement based on tolerances. The quality of the DEM was assessed by calculating the root mean square error (RMSE) (Chaplot et al., 2006), and by comparing the derived flow accumulation map with the actual stream patterns on the field. All major terrace failures coincided with this flow accumulation map, which indicated that the DEM gave a correct representation of the topography. To create the original topography before abandonment we adapted the GPS survey file by removing all points that represent the terrace failures and assuming that the original surface of the terraces was at the same level as undisturbed places in the middle of the terrace. With the same procedure we created a new DEM, based on 1141 points, which represented the topography at time of abandonment in 1984. The erosion rate was calculated by subtracting the two DEMs and multiplying with a soil bulk density value of 1.33 kg dm^{-3}.

4.4 Results and discussion

4.4.1 Erosion on abandoned fields

From the 58 abandoned fields 32 were classified as fields with moderate to severe erosion. Table 4.1 summarizes the average properties for abandoned fields with and without erosion and indicates if differences between the two groups are significant based on the Pearson's chi-square and the t-test. The significant (P<0.05) differences between the two groups, based on sufficient observations, are barley as previous land use, presence of terraces, and maximum slope. That steeper slopes increase the occurrence of erosion is obvious, but the presence of terraces as risk factor for erosion seems contradictory. However, the presence of terraces implies that the field is located on sloping land and in addition the terrace topography marks steep gradients at the terrace walls, which makes them vulnerable for gully erosion and piping, especially after abandonment. The last significant factor was barley as previous land use. An explanation can be the negative land management of cereal cultivation with high soil losses (Lasanta et al., 2000), which degraded the soil already before abandonment. We also observed that soil crusts under barley are often stronger compared to crusts on other cultivated fields, which is probably related to the frequency of ploughing. Surprisingly potential drainage area appeared not to influence the occurrence of erosion on abandoned fields, as would be expected (Vandekerckhove et al. 2000). A possible explanation can be inaccuracies in the DEM, because potential drainage area was only determined in GIS. On the other hand runoff is not only a function of drainage area but also land use and substrate in the upstream catchment determine the amount of runoff and thus the occurrence of erosion.

Table 4.1. Comparison of properties of abandoned fields with and without erosion

Properties	**No erosion**	**Erosion**	***P*-value***
Number of fields	26	32	
Previous land use (number of fields)			
Barley	8	19	**0.030**
Orchard	14	13	0.315
Vineyard	4	0	*0.021*
Parent material (number of fields)			
Marl	11	16	0.559
Colluvium	6	13	0.157
Keuper	4	2	*0.256*
Sandstone	3	1	*0.209*
Limestone	2	0	*0.110*
Vegetation (number of fields)			
Mainly herbs	12	12	0.506
Mixed herbs, grasses and shrubs	9	18	0.100
Mainly shrubs	5	2	*0.131*
Field properties			
Surface (ha)	2.6	2.0	0.450
Age of abandonment (year)	9.9	9.2	0.712
Fields with terraces	7	21	**0.003**
Fields with earth dams	7	10	0.719
Mean altitude (m)	389	373	0.230
Maximum slope (degrees)	13.8	18.1	**0.016**
Mean slope (degrees)	5.1	5.7	0.288
Potential drainage area (ha)	15.3	13.5	0.829
Mean solar radiation (MJ cm^{-2} $year^{-1}$)	0.82	0.81	0.769

* Bold values indicate significant differences (P<0.05) as determined with the t-test and Chi-square test. P values in italic had too few observations to be reliably significant

4.4.2 *Terrace failure*

Since the presence of terraces appeared to be the most significant factor for erosion on abandoned fields we studied the occurrence of terrace failure in more detail. Table 4.2 shows the results of the statistical analysis of terrace failure for all terraces and for abandoned terraces only. From the 288 terraces that were included in this study 121 terraces were collapsed. The results for all terraces showed that abandoned land, valley bottom position, slope of terrace, time since ploughing, a loam texture, total vegetation cover, grasses on terrace and shrubs on terrace wall had a significant positive relationship with terrace failure, while orchards, hillslope position, infiltration, trees on terrace and grasses on terrace wall had a significant negative relationship with terrace failure. Land abandonment and its related factors, like time since ploughing and vegetation cover, were the most significant variables to induce terrace failure, more than the topographical and soil variables. This means that land abandonment indeed increases the risk of terrace failure and subsequently enhances erosion on abandoned fields.

When we applied the same analysis for just the abandoned terraces, more topographical and physical variables became important, like channel position, potential drainage area, height of terrace wall, and Quaternary colluvium. Quaternary colluvium, channel position, potential drainage area, loam texture, trees on terrace, height terrace wall and shrubs on terrace wall had a significant positive relationship with terrace failure, while sandy loam texture, and infiltration had a significant negative relationship with terrace failure for the abandoned terraces. One of the interesting results with respect to mitigation of terrace failure was the variable shrub cover on terrace wall, which significantly increased terrace failure, while grasses had a stabilising effect on terrace failure. This positive stabilising effect of grasses was also experimentally found by De Baets et al. (2006). This result should be taken into account in revegetation schemes for soil and water conservation.

Besides the analysis of which factors induce terrace failure, we also constructed two logistic regression models to determine the probability of terrace failure. For the first model we used all variables, which resulted in a consistent regression model of 10 variables, using both forward and backward stepwise logistic regression (Table 4.3). The Hosmer-Lemeshow test resulted in a correct prediction percentage of 74.9 and the ROC value was 0.800, which indicates that the model performs quite well. The standardized beta-coefficients indicate that none of the variables has a very strong or weak influence on the predicted probability, which means that terrace failure is determined by a combination of factors and not a single variable. Using only the spatial variables, in order to predict the spatial distribution of terrace failure, the logistic regression analysis resulted in a regression model with six variables. However, the ROC value was 0.736 and the result of the Hosmer-Lemeshow test gave an overall correct prediction percentage of 70.8, which is lower than the values of the regression model that uses all variables. This means that exclusion of variables for which no spatial distribution data is available, e.g. time since ploughing, decreases the performance of the regression model.

Table 4.2. Comparison of properties of intact and failed terraces for all terraces and for abandoned terraces only

Properties	**All terraces**			**Abandoned terraces**		
	Intact	**Failure**	***P*-value***	**Intact**	**Failure**	***P*-value***
Number of terraces	167	121		76	92	
Land use						
Barley	19	10	0.386	-	-	-
Orchard	72	19	**0.000**	-	-	-
Abandoned land	76	92	**0.000**	-	-	-
Geology						
Quaternary colluvium	40	41	0.064	9	30	**0.002**
Tertiary marl	63	42	0.600	35	33	0.181
Cretaceous marl	64	38	0.226	32	29	0.156
Topography						
Hillslope	85	39	**0.002**	30	27	0.168
Valley bottom	61	67	**0.001**	44	51	0.749
Channel	21	15	0.964	2	14	**0.006**
Terrace properties						
Altitude (m)	384	391	0.152	391	391	0.961
Natural slope (degrees)	4.2	3.9	0.259	3.6	3.9	0.408
Slope of terrace (degrees)	0.7	0.9	**0.044**	0.76	0.82	0.606
Radiation (MJ cm^{-2} $year^{-1}$)	0.8	0.8	0.629	0.81	0.8	0.116
Potential drainage area (ha)	6.9	9.1	0.111	4.9	10	**0.001**
Time since ploughing (class 1-7)	2.7	3.8	**0.000**	4.4	4.5	0.558
Soil properties						
Silt loam texture	115	71	0.074	53	52	0.078
Loam texture	36	45	**0.004**	10	35	**0.000**
Sandy loam texture	16	5	0.079	13	5	**0.015**
Aggregate stability	10.7	10.2	0.725	10.9	10	0.529
EC (μS cm^{-1})	392	425	0.828	336	441	0.596
Infiltration (class 1-3)	1.81	1.60	**0.005**	1.62	1.43	**0.044**
Terrace cover						
Rock fragment cover (class 1-4)	1.98	1.88	0.402	1.7	1.84	0.308
Total vegetation cover (%)	36.6	48.3	**0.000**	54.1	56.4	0.388
Trees (%)	7.7	5.2	**0.014**	2.53	4.91	**0.013**
Shrubs (%)	1.0	2.1	0.087	2.09	2.68	0.569
Grasses (%)	29.9	43.7	**0.000**	50.7	52.0	0.678
Terrace wall properties						
Height terrace wall (m)	1.75	1.75	0.973	1.53	1.75	**0.032**
Width terrace wall (m)	1.9	1.96	0.502	1.76	1.90	0.178
Slope terrace wall (degrees)	43.0	41.7	0.129	41.4	42.7	0.242
Trees (%)	1.7	1.0	0.284	0.5	0.6	0.939
Shrubs (%)	5.3	9.8	**0.010**	1.9	11	**0.000**
Grasses (%)	45.3	34.7	**0.000**	32.8	31.6	0.686
Maintenance	8	5	0.791	-	-	-
Stone consolidation	11	14	0.138	5	10	0.332

* Bold values indicate significant differences ($P<0.05$) as determined with the t-test and Chi-square test

Table 4.3. Standardized logistic regression coefficients and the performance of the two models

Variable	All variables	Spatial variables
Abandoned land		0.170
Orchard	-0.218	-0.235
Quaternary colluvium	0.261	0.219
Valley bottom	0.214	
Altitude		0.133
Slope of terrace	0.113	
Radiation	-0.190	-0.175
Potential drainage area	0.185	0.226
Time since ploughing	0.162	
Loam texture	0.189	
Slope terrace wall	-0.116	
Shrub cover terrace wall	0.146	
ROC	0.800	0.736
Percentage correct predicted*	74.9	70.8

* Based on the Hosmer-Lemeshow test

The results showed that terrace failure is frequently occurring and that land abandonment is an important driver of terrace failure. Only few other studies have addressed terrace failure. Lasanta et al. (2001) studied the collapse of terrace walls due to abandonment of the traditional land management in a mountainous area in northern Spain. The average size of the terrace collapse was 3.3 m^3 and the found positive correlations between the volume of removed material and the height of the terrace wall and the slope gradient. Romero Díaz et al. (2007) showed that a landscape in Southeast Spain, which was terraced in the 1970s, is now completely abandoned and the terraces are destroyed by piping. Piping was mainly caused by the lack of soil structure and the dispersive character of the soil material. Due to agricultural EU policy that encouraged farmers to cultivate certain crops by subsidising their expanded cultivation on a ‘per hectare’ basis (Cots-Folch et al., 2006), the cultivation of almonds and grapes increased. Ramos et al. (2007) found that new constructed terraces for vineyards were not sustainable, with heights and widths that are greater than the accepted design criteria, since farmers only considered trafficability for terrace construction. This resulted in destruction of large parts of the terraces during an extreme rainfall event. When slopes are less steep land levelling occurs without the construction of terraces. This clearance of native Mediterranean vegetation for almond cultivation without erosion-prevention terraces resulted in serious gully erosion (Faulkner, 1995) and increased soil redistribution due to tillage erosion (Van Wesemael et al., 2006).

4.4.3 *Quantification of erosion*

Figure 4.3 shows the surrounding of the field site and the calculated sediment losses after subtracting the current DEM with the terrace failures from the 1984 DEM. The main terrace failures are clearly visible with incisions of more than one meter. Nevertheless some sedimentation has occurred as well, especially below the terrace walls and on a small alluvial fan from a large gully in the side wall. The average net surface lowering since abandonment was 13.8 cm, resulting in a net erosion rate of 87 ton ha^{-1} $year^{-1}$. This rate is higher than the average 12 ton ha^{-1} $year^{-1}$ calculated from several gully erosion studies (Poesen et al., 2003) and much higher than the usual range of 0.1-1 ton ha^{-1} $year^{-1}$ under semi-natural vegetation (Martínez-Fernández and Esteve, 2005) and even semi-arid badlands have generally lower erosion rates (Canton et al., 2001). Although the field site has a rather large drainage area (113 ha) and erodible soils, the main cause for the high erosion rates is probably the presence of terraces in combination with land abandonment. Due to the absence of tillage the storage capacity decreased and soil crusts started to develop, which increased overland flow and led to gully erosion (Imeson et al., 1998). As long as the terrace walls are maintained they serve as soil and water conservation structures, however, with time a large amount of sediment is stored behind the terraces, which forms a potential source of sediment that can be easily released after terrace failure.

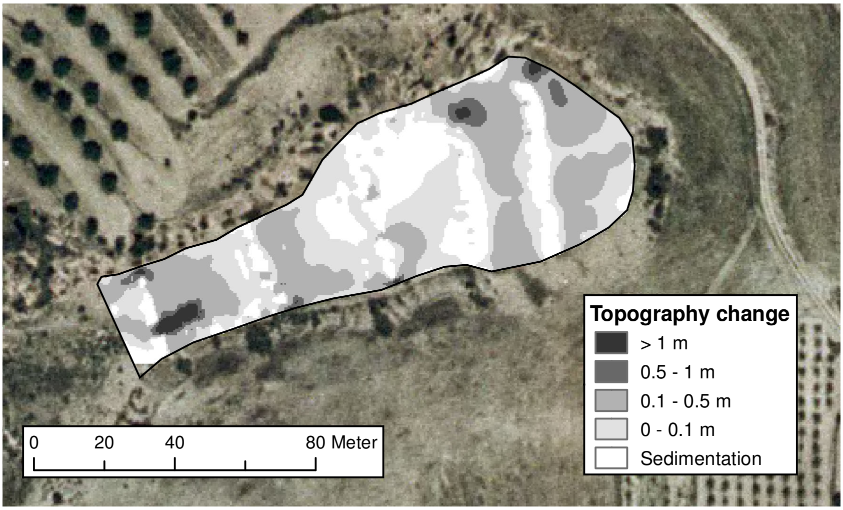


Figure 4.3. Sediment losses for the terrace field since abandonment in 1984

We based the calculation of the erosion rate on the subtraction of two DEMs. However, the DEMs were created by interpolation of GPS survey points, which may have introduced errors (Chaplot et al., 2006). Therefore we compared the results of our selected ‘topo to raster’ technique with some other interpolation methods (Table 4.4). The RMSE values, as calculated for the current DEM, were lowest for the DEM based on TIN (Triangulated Irregular Network) and the 'topo to raster' DEM. Between the different methods a considerable difference between the minimum and maximum topography change existed, but the calculated erosion rates were similar and were all higher than the reported gully erosion rates of Poesen et al. (2003). Only for the 'topo to raster' method that included the surrounding area the sediment delivery rate was higher, probably because the steep slopes at the border of the field influenced the interpolation. Although the DEM based on TIN had a slightly lower RMSE, we used the 'topo to raster' method since its secondary derivates such as slope and flow accumulation map were more realistic and the minimum and maximum altitude changes were closest to reality, considering the depth of 1.5 meter of the largest gully and no signs of much sediment accumulation.

Table 4.4. Comparison of different interpolation methods for DEM construction

Interpolation method	**RMSE** (m)	**Erosion rate** (ton ha^{-1} yr^{-1})	**Min. change** (m)	**Max. change** (m)
Topo to raster	0.166	87	-1.91	0.24
Kriging	0.265	96	-2.25	0.41
Spline	0.477	72	-2.77	3.32
DEM based on TIN	0.149	91	-1.82	0.61
Topo to raster incl. surrounding area	0.268	119	-2.29	0.70
Individual plane for each terrace	0.166	81	-1.95	0.93

4.4.4. Options for mitigation

The results of our study showed that erosion on abandoned fields is widespread and especially terrace failure is a major source of sediment. Soil and water conservation practices to mitigate soil erosion after agricultural land abandonment, and terrace failure in specific, can be divided into two groups: (1) maintenance of terrace walls and earth dams, and (2) revegetation with indigenous species. The first option should include restoration of terrace walls after heavy rainfall and also ploughing on the terrace can be an option to enhance storage capacity and improve infiltration. As a result more water will be retained on the terrace, which enhances vegetation growth and consequently decreases erosion. This kind of management might even include subsidies for farmers who combine extensive agriculture with soil and water conservation practices. For example, people from marginal rural areas in the province of Almería (Southeast Spain) receive subsidies when they remain working on the field and maintain agricultural terraces. These subsidies are intended to prevent depopulation of the rural areas and are paid from the EU less favoured areas support scheme. However, on the long term this option might not be very profitable, since the cost of subsidies will be high and when such a subsidy program stops the farmer might still abandon those fields, which means that erosion is only delayed. This view is confirmed by Oñate and Peco (2005), who interviewed relevant stakeholders in Southeast Spain about policies regarding desertification. Another aspect that has to be considered is the recurrence time of events, since a large event will even destroy part of the maintained terraces. Cammeraat (2004) found for a nearby area that an event with a return period of 10 years resulted in failure of terraces and earth dams. Nevertheless, direct repair of failed terraces will prevent further development of the gullies in the terrace walls.

For the second option of revegetation a distinction can be made between (i) revegetation on the terrace to improve infiltration and (ii) revegetation of terrace walls and zones with concentrated flow to prevent or mitigate gully erosion. The effectiveness of vegetation in the reduction of runoff and erosion has been widely published (e.g. Morgan, 1995; Quinton et al., 1997; Cerdà, 1997). Vegetation protects the soil with its canopy and roots, and the input of organic matter improves the soil and hydrological properties, i.e. infiltration, porosity and structure, which decrease runoff and erosion (Bochet et al., 1999). The rapid establishment of a good vegetation cover is a key step for effective mitigation of erosion and requires first short-lived species that can grow quickly and later the “late successional” shrubs can establish (Obando, 2002). The alternative of revegetation only on and near the terrace walls has hardly any consequences for future land use and may positively influence terrace stability as a result of increased topsoil cohesion by surface roots. Furthermore, the resistance against fine scale slope failure processes may increase by including species with deeper penetrating roots (Gray, 1995). Faulkner (2007) showed that revegetation can even decrease the risk of piping by lowering the SAR and EC values of the soil.

Gyssels et al. (2005) concluded that for splash and sheet erosion vegetation cover is the most important parameter to control erosion, but for rill and gully erosion the effect of plant roots is at least as important. De Baets et al. (2006) demonstrated that especially grass roots are very effective in reducing soil detachment rates under concentrated flow. An increase in root density from 0 to 4 kg m^{-3} already decreased the relative soil detachment rates to very low values. Our results also showed that grasses on the terrace wall had a significant stabilising effect on terrace failure, while the presence of shrubs increased the risk of terrace failure. Hence, indigenous grass species with a dense rooting system and good vegetation cover are most suitable for revegetation of terrace walls. For our study area *Lygeum spartum*, *Brachypodium retusum*, *Piptatherum miliaceum* and *Stipa tenecissima* among others accomplish these characteristics (De Baets et al., 2007). However, other characteristics as germination and growth rate are important as well for successful mitigation of terrace failure, but not much information is available for semi-natural vegetation. From an ecological point of view the re-establishment of the indigenous shrub vegetation is a key step in the restoration of abandoned agricultural semi-arid lands (Caravaca et al., 2003). However, the use of native grass species in concentrated flow zones and on terrace walls seems to be more effective to mitigate erosion. Quinton et al. (2002)

provide a list of indigenous Mediterranean species that can be used for the revegetation of abandoned land, including specific bioengineering properties.

A final comment has to be addressed to the balance between costs of conservation practices and costs of soil quality loss and offsite effects. For Mediterranean countries not much information is available, neither about the costs of conservation measures nor about the environmental costs of erosion. However, for proper implementation of conservation practices a basic cost-benefit analysis is a minimum requirement. A recent paper of Hein (2007) suggests that the costs of erosion for a catchment in Southeast Spain are limited. However, he only considered local costs due to loss of nutrients, but no off-site effects like reservoir sedimentation were taken into account. The relation between recurrence time of events, sustainability of terraces and economic costs remains still unclear and deserves therefore further attention. One of the ways to reduce costs of mitigation practices is to specifically target measures to hotspot areas in the landscape where erosion is a problem at present, or where, if improperly managed, it will become a significant problem. In the case of the Carcavo basin mitigation measures should be targeted to areas where concentrated flow near terrace walls is a problem.

4.5. Conclusion

Abandonment of agricultural land is widespread and increasing in Mediterranean countries and can potentially lead to a considerable increase in erosion in semi-arid environments. Especially abandoned terrace fields are vulnerable because of gully erosion through the terrace walls. In the Carcavo basin more than half of the abandoned fields have moderate to severe erosion and the calculated erosion rate is high. Land abandonment, steeper terrace slope, loam texture, valley bottom position, and shrubs on the terrace wall are factors that increase the risk of terrace failure. This and several other studies show that terracing, although intended as conservation practice, actually enhances erosion, especially after abandonment. Construction of new terraces should therefore be carefully planned and being built according to sustainable design criteria and not in soils that are vulnerable to piping. To mitigate erosion on these abandoned fields the following soil and water conservation practices are possible: (1) maintenance of terrace walls in combination with increasing vegetation cover on the terrace, and (2) revegetation with indigenous grass species on zones with concentrated flow to prevent gully erosion. These practices should be specifically targeted to hotspot areas in the landscape where erosion is or will become a problem.

5. Upscaling fractional vegetation cover using QuickBird imagery and detailed aerial photographs in a semi-arid environment *

5.1 Introduction

Semi-arid lands cover 15 percent of the earth's land surface (Deichmann and Eklundh, 1991) and are characterized by low and erratic or seasonal rainfall. Vegetation in these areas is under stress due to limited availability of water, which results in a heterogeneous pattern of bare soil and vegetation patches (Valentin et al., 1999). These vegetation patterns are driven by the positive feedback between plant density and infiltration coupled to the redistribution of runoff (HilleRisLambers et al., 2001). The heterogeneity of the vegetation has important implications for biotic and abiotic processes, e.g. plant dynamics and biodiversity (Hutchings et al., 2000), and runoff and soil erosion (Puigdefabregas, 2005).

Several studies showed that vegetation cover and its spatial distribution are the key factor controlling overland flow generation (Thornes, 1990; Gutierrez and Hernandez, 1996; Quinton et al., 1997; Cerdà, 1998). Bare patches between vegetation function as runoff generating areas, which are generally bare rock and crusted areas that are characterised by poor soil structure with low infiltration rates. Whereas soils under vegetation have more organic matter, improved soil properties and higher infiltration capacity, which makes these patches sinks for runoff (Bergkamp, 1998; Cammeraat and Imeson, 1999). A threshold of 30 percent vegetation cover is already sufficient to decrease soil erosion considerably (Francis and Thornes, 1990; Rogers and Schumm, 1991; Quinton et al., 1997). Lesschen et al. (2008a) found a strong linear relation between fractional vegetation cover (FVC) and several spatial metrics that describe vegetation patterns on abandoned and semi-natural fields. Such a strong relationship was also found by Imeson and Prinsen (2004) for vegetation patterns of *Stipa tenacissima* shrublands. This means that FVC is a good proxy to describe spatial vegetation structures in landscapes with spotted vegetation patterns and, in addition, a good indicator of runoff and erosion risk. Knowledge about the spatial distribution and density of vegetation cover at the catchment scale is therefore essential to improve runoff and erosion predictions.

* Submitted as Lesschen, J.P., Cammeraat, L.H., Seijmonsbergen, A.C. and Hooke, J.M. Upscaling fractional vegetation cover using QuickBird imagery and detailed aerial photographs in a semi-arid environment. International Journal of Remote Sensing

Most studies that estimate vegetation cover from remotely sensed images use a spectral mixing model with two endmembers based on the NDVI (Wittich and Hansing, 1995; Baret et al., 1995; Carlson and Ripley, 1997; Gutman and Ignatov, 1998). Spectral mixing modelling divides each ground resolution element into its constituent materials using endmembers which represent the spectral characteristics of the cover types (Gilabert et al., 2000). Other studies use more complex spectral mixture models with more endmembers to estimate vegetation cover (Elmore et al., 2000; Xiao and Moody, 2005). However, most of the vegetation indices have been developed in temperate zones, which generally have a more homogeneous vegetation cover. Besides, all these studies were based on low or medium resolution satellite sensors, e.g. AVHRR, MODIS, and Landsat, while the current availability of high-resolution satellites, such as IKONOS and QuickBird, allows for more detailed studies of heterogeneous vegetation patterns. Besides, high resolution remote sensing data is essential for accurate assessment of erosion hotspots, since the processes that control erosion operate at the plot scale. Satellite remote sensing has been frequently used for assessment of erosion at the regional scale, but almost all based on low or medium resolution satellite images (Vrieling, 2006). Quincey et al. (2007) used a QuickBird image for the classification of land cover for erosion modelling in a small Himalayan catchment.

The objective of our study is to evaluate which vegetation index is most suitable for upscaling fractional vegetation cover in a semi-arid environment using a high resolution QuickBird image. At plot scale we made detailed aerial photographs, which were classified into bare and vegetated patches to derive the fractional vegetation cover. For these plots we calculated different vegetation indices, which were derived from a QuickBird image. Finally, the calculated indices were compared with the observed fractional vegetation cover to test which vegetation index had the best fit and is most appropriate to upscale FVC to the entire study area.

5.2 Methodology

5.2.1 Study area

The Carcavo basin is located in Southeast Spain about 40 km northwest of the city of Murcia, near the town of Cieza (Figure 5.1). It is a small catchment of 30 km^2 with altitudes ranging between 220 and 850 metres. This region of Spain is one of the driest in Europe with an average annual rainfall of 300 mm and a potential evapotranspiration of 900 mm. The geology of the area consists of steep Jurassic carbonate mountains with calcareous piedmonts, Keuper gypsum marls and basin deposits of Cretaceous and Miocene marls. Current land use in the study area consists of barley, olive and almond orchards, vineyards, abandoned land, reforested land, shrubland and forest. Just north of the study area irrigated peach orchards are found in the valley of the Segura River. In the 1970s large parts of the catchment were planted with pine (*Pinus halepensis*) within the framework of reforestation and soil conservation programs. During the last few decades parts of the non-irrigated agriculture have been abandoned and are currently under different stages of secondary succession. The semi-natural vegetation on slopes is mainly composed of *Rhamno lycioidis-Quercetum cocciferae* shrubland in upper areas, while *Stipa tenacissima* and dwarf-shrubs communities are dominant in lower areas. All vegetation in the study area, except for some forest areas on steep north slopes, is characterised by a heterogeneous pattern of bare soil and vegetation patches.

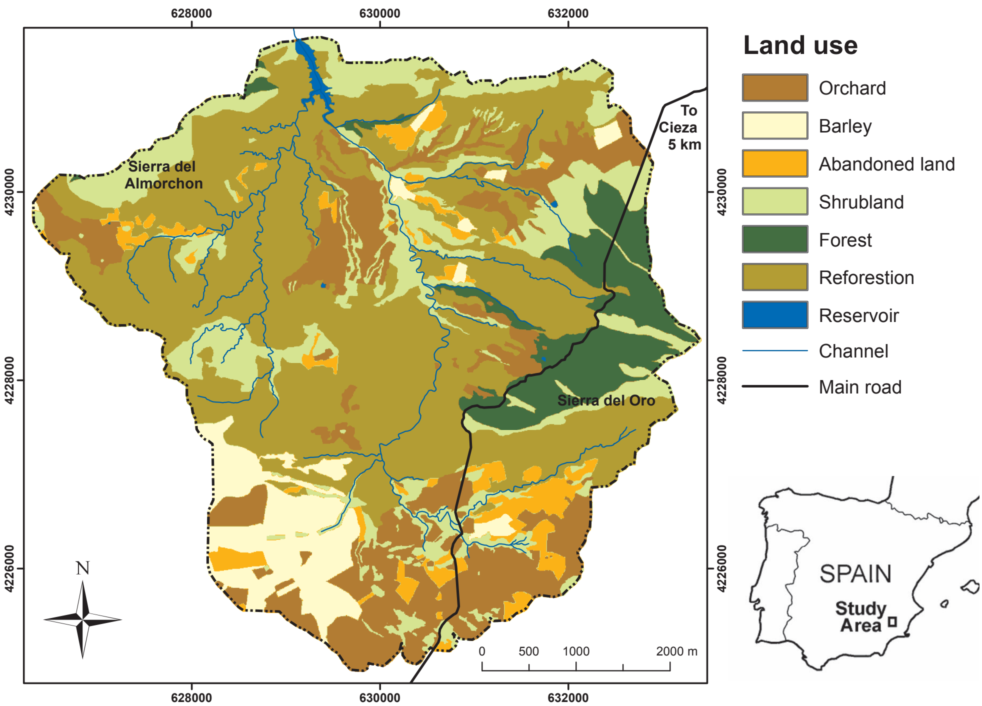


Figure 5.1. Location of the Carcavo basin and its land use

5.2.2 Detailed aerial photographs

To determine FVC of different land uses we used detailed aerial photographs. Conventional aerial photographs have resolutions of about 1 metre, which is too coarse to clearly distinguish the bare and vegetated patches. Therefore, we used a balloon-mounted camera system (Ries and Marzolff, 2003) to make detailed aerial photographs of the vegetation cover on each plot. The photos were made in April 2006 with a Canon Digital IXUS 500 camera attached to a large balloon filled with helium (see also Lesschen et al., 2008a). With a radiographic device the camera was activated, which allowed us to take detailed aerial photographs from 30-50 metres above the surface. These images had a spatial resolution of 1-2 centimetres, which made them suitable for determination of the fractional vegetation cover. We selected 20 representative plots with different land uses and different fractions of vegetation cover. For each plot we extracted a representative part of 10 by 10 metre from the central part of the digital photo, to avoid distortion from the photo margins. All images were resized to a pixel resolution of 2 cm. Next, the images were classified into bare and vegetated patches. For the classification we examined several default classification methods within the remote sensing package ENVI 3.5 (Research Systems, Boulder). The supervised maximum likelihood method performed best, since it classified vegetation areas most accurately and it was the only method that classified shaded areas correctly. The maximum likelihood method assigns each pixel to the class with the highest probability and evaluates both the variance and covariance of the spectral response pattern (Lillesand et al., 2004).

5.2.3. QuickBird image processing

The QuickBird image was taken on 25 June 2006 at 13:21 hours for an area of 9 by 10 km, which included the entire Carcavo basin. Because of the date and time of acquisition the image was not disturbed by shaded areas. The orthorectified 4 band multi-spectral QuickBird image was used for the analysis. This image has a blue (B), green (G), red (R) and near-infrared (NIR) band with a resolution of 2.8 metre. The different bands of the QuickBird image were converted to satellite reflectance, which is the input for most vegetation indices. Therefore, we first calculated the satellite radiance by multiplying the QuickBird bands with their absolute radiometric calibration factors and dividing them by the effective bandwidth. Next, the satellite reflectance was calculated for each band using the following equation:

$$\rho = \frac{\pi \cdot L \cdot d^2}{Esun \cdot \cos(SZ)} \tag{5.1}$$

where ρ is the satellite reflectance (range 0-1), L the satellite radiance (W m^{-2} ster^{-1} µm^{-1}), d the earth-sun distance in astronomical units (1.016485 for 25 June), *Esun* the mean solar irradiance (W m^{-2} µm^{-1}) and SZ the sun zenith angle in degrees (Krause, 2005).

Table 5.1. Selected vegetation indices and equations to calculate fractional vegetation cover

Vegetation index	Equation	Reference
Ratio Vegetation Index	$RVI = \frac{NIR}{R}$	Jordan (1969)
Normalized Difference Vegetation Index	$NDVI = \frac{NIR - R}{NIR + R}$	Rouse et al. (1973)
Weighted Difference Vegetation Index	$WDVI = NIR - aR$, where a is the slope of the soil line	Clevers (1989)
Soil Adjusted Vegetation Index	$SAVI = \left(\frac{NIR - R}{NIR + R + L}\right) \times (1 + L)$, where L is a soil adjustment factor, ranging between 0-1	Huete (1988)
Modified Soil Adjusted Vegetation Index 2	$MSAVI_2 = \frac{2NIR + 1 - \sqrt{(2NIR + 1)^2 - 8(NIR - R)}}{2}$	Qi et al. (1994)
Visible Atmospherically Resistant Index	$VARI = \frac{G - R}{G + R - B}$	Gitelson et al. (2002)
Canopy gap fraction	$FVC_{Baret} = 1 - \left(\frac{NDVI - NDVI_{\infty}}{NDVI_S - NDVI_{\infty}}\right)^{0.6175}$, where $NDVI_{\infty}$ is the NDVI for complete vegetation cover and $NDVI_S$ the NDVI for bare soil	Baret et al. (1995)
Fractional vegetation cover	$FVC_{Carlson} = \left(\frac{NDVI - NDVI_S}{NDVI_{\infty} - NDVI_S}\right)^2$	Carlson and Ripley (1997)
Green vegetation fraction	$FVC_{Gutman} = \frac{NDVI - NDVI_S}{NDVI_{\infty} - NDVI_S}$	Gutman and Ignatov (1998)
Scaled Difference Vegetation Index	$SDVI = \frac{NIR - R - (NIR_S - R_S)}{NIR_V - R_V - (NIR_S - R_S)}$, where S denotes the reflectance of bare soil and V the reflectance of complete vegetation cover	Jiang et al. (2006)

The locations of the selected plots were digitised on the QuickBird image in ArcGIS, using the pan-sharpened image. This image had a higher resolution (0.7 m), which made it possible to accurately identify the locations of the plots. For each plot the average reflectance of each band was calculated using a rectangular neighbourhood function of four grid cells. We selected six vegetation indices and four fractional vegetation cover equations (Table 5.1) to link FVC of the plots to the QuickBird image. The selection of the vegetation indices was based on the three categories as distinguished by Rondeaux et al. (1996): intrinsic indices, soil-line related indices and atmospheric-corrected indices. We selected the well-known and often used vegetation indices of each category. In addition, we applied a stepwise linear regression analysis using the four spectral bands of the QuickBird image. To compare how well each index related to the observed FVC we used linear regression.

5.2.4. Vegetation indices

The Ratio Vegetation Index (RVI) of Jordan (1969) is the simplest index, expressed as the ratio between the near-infrared and red reflectance. The most well know vegetation index is the Normalized Difference Vegetation Index (NDVI), which is the ratio between the difference and sum of the near-infrared and red reflectance (Rouse et al., 1973). The Weighted Difference Vegetation Index (WDVI) corrects for the soil background by multiplying the red reflectance with the slope of the soil line (Clevers, 1989). The Soil Adjusted Vegetation Index (SAVI) is similar to the NDVI, but uses an adjustment factor L to correct for the soil background. The value of L ranges from 0, for very high vegetation cover, to 1 for very low vegetation cover. For our study we used an L-value of 0.5, suggested for intermediate vegetation densities (Huete, 1988). The $MSAVI_2$ is a modified version of the SAVI, for which no vegetation density dependent constant is required (Qi et al., 1994). The last vegetation index is the Visible Atmospherically Resistant Index (VARI), which is less sensitive to atmospheric effects, since only the bands in the visible spectrum are used (Gitelson et al., 2002).

The fractional vegetation cover equations from Baret et al. (1995), Carlson and Ripley (1997) and Gutman and Ignatov (1998) are similar and all based on the ratio of differences between observed NDVI values and NDVI values for bare soil and complete vegetation cover. The last equation is the Scaled Difference Vegetation Index (SDVI), which is in value the same as FVC, and is a scale invariant index (Jiang et al., 2006). For these last equations information about the spectral properties of bare soil and complete vegetation cover is required. Therefore we selected 30 points on bare soil scattered over the catchment and 30 points on irrigated peach orchards, which represent the densest vegetation in the area. For these points the reflectances were derived from the QuickBird image and the NDVI values were calculated.

5.3. Results

For the 20 classified images the FVC ranged from 0.01 for almost bare soil to 0.73 for a plot with riparian vegetation. The FVC for almond fields ranged from 0.08 to 0.22, for abandoned fields from 0.32 to 0.49, for shrublands from 0.33 to 0.57 and for reforested plots from 0.46 to 0.63. Examples of the plots and their classified images and FVC are shown in Figure 5.2. Besides the calculation of the vegetation indices, we also applied a stepwise linear regression using the reflectance of the four QuickBird bands, which resulted in the following regression model:

$$FVC = 9.81 \times G - 9.69 \times R + 0.641 \qquad (5.2)$$

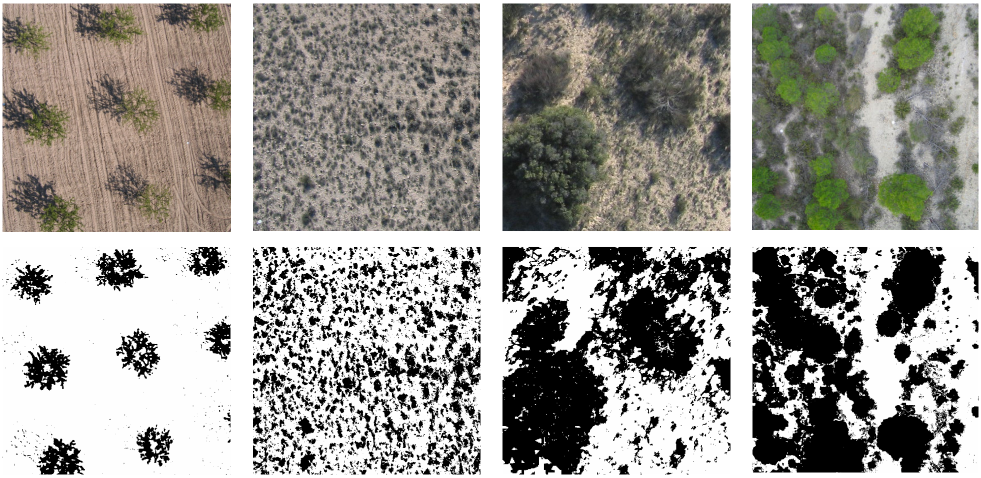


Figure 5.2. Examples of detailed aerial photos of different land uses with their classified images and FVC

The results of the comparison of the observed fractional vegetation cover and vegetation indices, based on linear regression, are presented in Table 5.2. The regression based on the reflectance of the QuickBird bands had the best fit with a R^2 of 0.91. The VARI, which is also based on bands in the visible spectrum, had still a good fit with a R^2 of 0.78. The RVI, NDVI, FVC_{Baret} and FVC_{Gutman} had reasonable fits with R^2 around 0.68. All other indices had only a weak or even no relation with the observed FVC with R^2 below 0.5. Although the equations that calculate the fractional vegetation cover directly had reasonable fits with the observed FVC, the absolute values were much lower than the observed FVC, especially for the plots with high vegetation cover. The mean difference between the observed FVC and calculated FVC_{Baret}, $FVC_{Carlson}$ and FVC_{Gutman} was respectively 0.29, 0.35 and 0.25.

Using the regression equation based on the green and red band, which had the highest R^2, we created a map of the fractional vegetation cover for the entire study area (Figure 5.3). The map clearly shows that the forest areas, which are mainly located on north slopes, have an FVC of more than 0.7, while most agricultural lands have a FVC between 0.0 and 0.3. To check the FVC map we also looked at which percentage of all pixels was outside the 0-1 range. Of all pixels 4.2 percent had FVC values below zero, which were locations with very strong soil brightness. However, all these locations were bare soil, mainly from recently ploughed fields, which were therefore reclassified to zero vegetation cover. Only 0.065 percent of all pixels had FVC values above one, however, all these pixels where located at sites with open water, which caused the atypical reflectance.

Table 5.2. Relationship between observed and predicted FVC (*n*=20)

Index	Regression equation	R^2	*P* value
RVI	FVC = 1.26 × RVI - 1.38	0.66	0.000
NDVI	FVC = 3.93 × NDVI - 0.268	0.69	0.000
WDVI	FVC = 6.06 × WDVI + 0.0701	0.16	0.085
$SAVI_{0.5}$	FVC = 5.45 × SAVI - 0.239	0.31	0.011
$SAVI_{0.8}$	FVC = 4.93 × SAVI - 0.132	0.19	0.054
$MSAVI_2$	FVC = 4.57 × $MSAVI_2$ - 0.0864	0.16	0.078
VARI	FVC = 3.01 × VARI + 0.762	0.78	0.000
FVC_{Baret}	FVC = 3.61 × FVC_{Baret} + 0.0973	0.68	0.000
$FVC_{Carlson}$	FVC = 6.11 × $FVC_{Carlson}$ + 0.253	0.49	0.001
FVC_{Gutman}	FVC = 2.40 × FVC_{Gutman} + 0.0857	0.69	0.000
SDVI	FVC = -0.215 × SDVI + 0.370	0.00	0.777
$FVC_{regression}$	FVC = 9.81 × G - 9.69 × R + 0.641	0.91	0.000

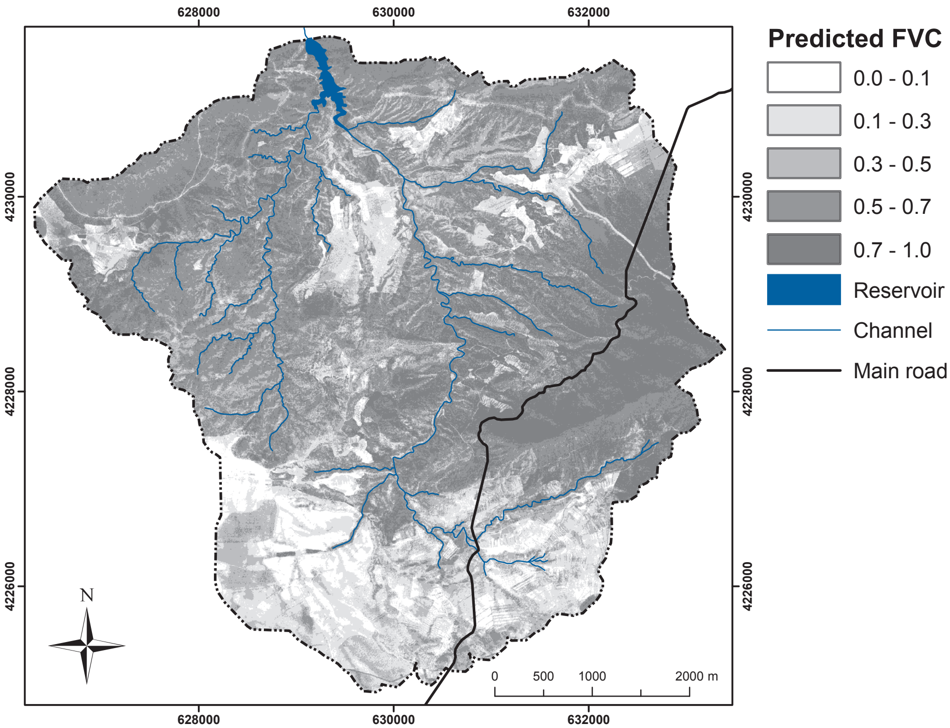


Figure 5.3. Predicted FVC map for the Carcavo basin

5.4. Discussion

We found that a simple regression model based on the green and red reflectance had the best fit with observed FVC, which makes this regression model the most suitable vegetation index for upscaling FVC. Our results showed that vegetation indices based on the difference between the NIR and red reflectance, such as the WDVI and SDVI, had the lowest correlation with vegetation cover, while VARI and our regression model, which are based on bands in the visible spectra, had the highest correlation with vegetation cover.

The reason for the low fit of the DVI based vegetation indices is illustrated in Figure 5.4, which shows the red and NIR reflectances of randomly selected pixels from the QuickBird image for different land uses. Normally a low vegetation cover would be expected for points near the soil line, while points with dense vegetation cover should have high NIR reflectance and low red reflectance (Jiang et al., 2006). However, this is not the case for the semi-arid landscape of the Carcavo basin under the conditions when the QuickBird image was taken. Barley fields still show the expected spectral signature, with high red and NIR reflectances near the soil line, since these fields were already harvested before the acquisition date of the QuickBird image. Peach orchards also show the typical spectra with high NIR reflectance and low red reflectance, however, forest has low red and NIR reflectance, while both land uses have a similar dense vegetation cover. Differences in plant physiology and morphology, which affect the spectral behaviour, are the most likely explanation. Vegetation of arid and semi-arid areas is usually adapted to survive high temperature and low water availability, e.g. small leaves, changed leaf angle, wax coatings and reduced photosynthetic activity (Calvão and Palmeirim, 2004). These adaptations influence its detectability by remote sensing techniques, due to lack of a strong red edge, reduced leaf absorption in the visible spectra, and strong wax absorptions (Okin et al., 2001). The forest and shrubland vegetation of the Carcavo basin is adapted to these dry conditions, whereas peach is irrigated and therefore not physiological adapted. Sandholt et al. (2002) also found that NDVI values may vary depending on water limitation, which can lead to underestimation of FVC. Vegetation cover in semi-arid areas appears to be more related to differences in intensity of the reflectance than to differences between the spectral bands. Vegetation indices based on the difference between the NIR and red reflectance, like the NDVI, are therefore not the most appropriate indices for FVC determination in semi-arid environments.

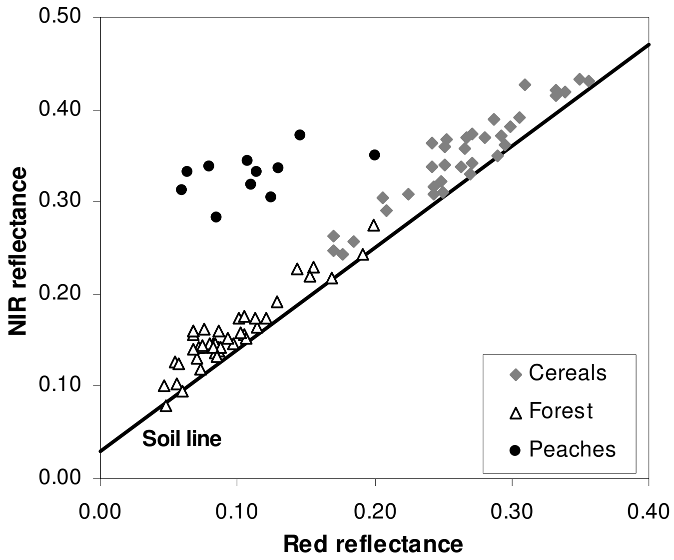


Figure 5.4. Plot of the red and NIR reflectance for different land uses

Other studies also describe the atypical behaviour of the NIR reflectance under semi-arid conditions. Xiao and Moody (2005) found a differences between the spectra of green woody vegetation and green grass cover, particularly in the near-infrared (NIR) wavelength. Hurcom et al. (1996) determined spectral response curves for different species in semi-arid environments. These curves were very similar in the visible spectra, but the reflectance varied more in the NIR, which was attributed to wax coatings of some species, which increased the NIR reflectance. In addition, Gitelson et al. (2002) found that the reflectance of wheat fields in the NIR at the midseason decreased, which can be a limiting factor in the use of that spectral region for vegetation cover estimation.

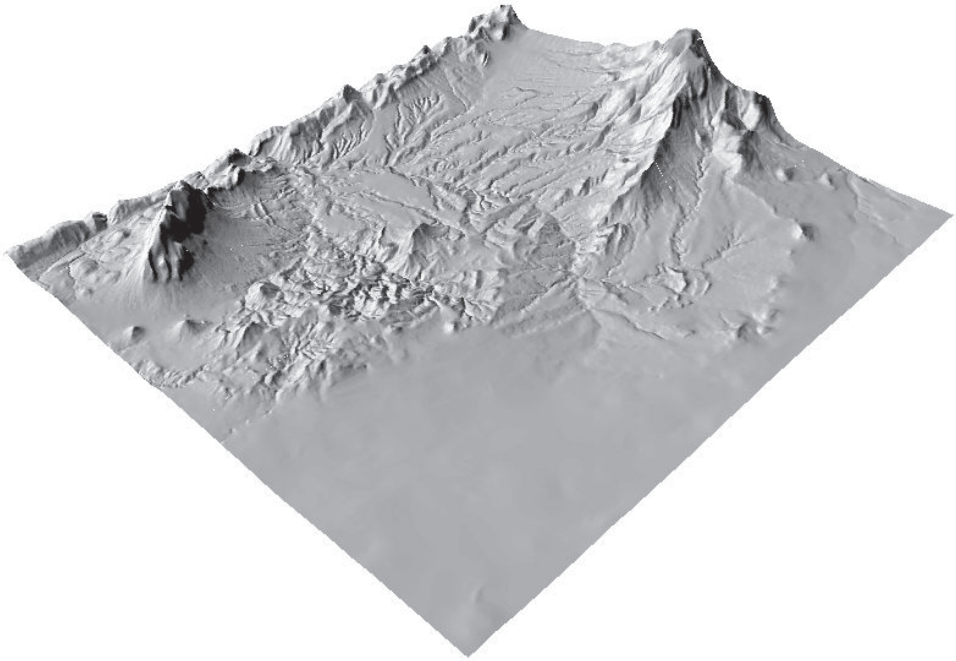
In our study the relationship between vegetation cover and most vegetation indices was linear. However, in literature no consistent relationship is found, since several studies find linear and others non-linear relationships between vegetation indices and vegetation cover (Xiao and Moody, 2005). This discrepancy is probably related to different ranges of FVC in the various studies, since NDVI becomes non-linear above a certain level of FVC, due to an increase of the leaf area index of the vegetation (Carlson and Ripley, 1997). However, in semi-arid environments a linear relationship between FVC and vegetation indices can be presumed, because of low vegetation cover.

Another issue is the presence of biological soil crusts in semi-arid areas, which are formed by mosses, lichens and algae. In our study we did not include the biological soil crusts as separate class, but classified them as bare areas, despite the fact that the spectral reflectance of these microphytes under certain conditions can be similar to those of higher plants (Karnieli et al., 1996), which could lead to misinterpretation of the vegetation cover. However, the different spectral reflectance only occurs when the soil crusts are wet, whereas the spectral reflectance under dry conditions is similar to those of bare soil. Since our QuickBird image was acquired during a dry period, we think that the biological soil crusts did not affect our vegetation cover estimates.

The obtained FVC map is very useful for the assessment of erosion risk. Combining the before mentioned threshold of 30% vegetation cover with a simple erosion model, which takes topography, geology and land use into account, enables a quick identification of the erosion hotspots within a catchment. In addition, the high resolution of the QuickBird image enables identification of areas with high erosion risk at the scale at which the erosion processes occur. Conservation practices such as revegetation can then be directly applied at locations with high erosion risk and low vegetation cover. Besides erosion risk assessment FVC maps are also valuable for other purposes, e.g. monitoring of changes in vegetation cover as indicator for desertification (Schlesinger et al., 1990; Pickup et al., 1993; Kefi et al., 2007), or as input for vegetation-climate models (Betts et al., 1997; Pitman, 2003).

5.5. Conclusion

Our study showed that the relationship between FVC and vegetation indices in a semi-arid environment with heterogeneous vegetation patterns differed strongly. A simple regression of the green and red reflectance appeared to be the best equation to upscale FVC in our study area. However, other indices had lower fits with observed FVC, especially the DVI based indices. The use of vegetation indices for determination of FVC in semi-arid areas without taking any calibration into account is therefore a very risky approach. Calibration based on local field data, e.g. from detailed aerial photographs or vegetation cover estimates from georeferenced plots, remains therefore essential for correct estimates of fractional vegetation cover.



6. Modelling runoff and erosion for a semi-arid catchment based on hydrological connectivity to integrate plot and hillslope scale influences *

6.1. Introduction

Scale dependency in erosion research has often been addressed as an important issue that deserves further attention (e.g. Poesen et al., 2003; Boardman, 2006). Simple extrapolations of plot measurements generally lead to large overestimations of runoff and erosion rates at catchment level, because of the non-linearity of runoff and erosion processes. Several studies demonstrated scale dependency with field experiments at different scales (e.g. Le Bissonnais et al., 1998; Cammeraat, 2002; Wilcox et al., 2003; Cerdan et al., 2004; Yair and Raz-Yassif, 2004). These studies showed a strong decrease in runoff and erosion rates with increasing plot length. Also the threshold for the occurrence of runoff increases at higher scale levels (Cammeraat, 2004). The overestimation of runoff and erosion rates can be attributed to the spatially varying influences of sinks, i.e. areas of infiltration and sedimentation. For example, plot scale studies of runoff and erosion do not reveal sediment deposition at footslopes or increased infiltration on agricultural terraces. However, runoff and erosion models mostly focus on one specific spatial scale level, i.e. field scale or catchment scale (Jetten et al., 1999), without addressing the influence of relevant sinks at other scales. Although few conceptual models have been developed that address this scale dependency in erosion simulation (Parsons et al. 2004; Lu et al. 2005), no erosion modelling study has actually addressed this scale dependency for real catchments.

It is well known that area specific runoff yield decreases with increasing area. For large watersheds this decrease is attributed to factors such as rain cell size, lateral changes in lithology and channel width and increasing storage possibilities in the valley domain. While for small areas the non-uniform infiltration and spatial variability of the vegetation and surface properties are responsible for the decrease in runoff rates (Yair and Raz-Yassif, 2004; Kirkby et al., 2005). Besides, Wainwright and Parsons (2002) and Reaney et al. (2007) demonstrated that temporal variations in rainfall intensity during a storm event can also lead to scale dependency in runoff. However, for erosion this relationship is less clear. Parsons et al. (2006) experimentally demonstrated that sediment yield decreases with plot

* Submitted as Lesschen, J.P., Schoorl, J.M. and Cammeraat, L.H. Modelling runoff and erosion for a semi-arid catchment based on hydrological connectivity to integrate plot and hillslope scale influences. Geomorphology.

lengths above 7 meter, due to the limited travel distance of individual entrained particles and by the decline in runoff coefficient as plot length increases. However, at broader scales additional erosion processes such as gully erosion, mass movement and bank erosion can become active and increase the area specific sediment yield. Nevertheless, from a certain basin area threshold, sediment yield becomes dominated by sediment transport and sediment deposition rather than by active erosion processes, and will consequently decrease with increasing basin area (De Vente and Poesen, 2005).

The concept of hydrological connectivity is becoming increasingly applied within the field of hydrology and geomorphology (Hooke, 2006; Bracken and Croke, 2007). Hydrological connectivity can be defined as the physical linkage of water and sediment through the fluvial system (Hooke, 2003). This definition is scale independent, which makes the concept of hydrological connectivity useful to take account of scale dependency in soil erosion research. Since hydrological connectivity describes the linkage of runoff and sediment, it finally determines whether runoff and sediment will become connected at broader scales. Identification of the areas that function as a sink is therefore crucial to model runoff and erosion at the catchment scale. The connectivity of runoff and sediment are not always linked, especially not in case of resistant substrates such as limestone, but also biological soil crusts can lead to high runoff rates but low erosion rates (Belnap, 2006). Although the term hydrological connectivity is also used for subsurface flow (Buttle et al., 2004; Ocampo et al., 2006), we only consider Hortonian overland flow in this paper, which is the main runoff generating mechanism in semi-arid environments (Bryan and Yair, 1982).

The main factors that, given a certain rainfall event, influence the hydrological connectivity at plot scale are micro-topography and vegetation. Vegetation in semi-arid ecosystems is characterised by a heterogeneous pattern of bare and vegetated patches, which make overland flow highly discontinuous owing to the non-uniform infiltration (Cerdà, 1998; Ludwig et al., 2005). Vegetation is also at hillslope scale one of the main factors influencing this connectivity. The spatial structure of these vegetation patterns finally determines the hydrological connectivity at the hillslope (Puigdefabregas et al., 1999). Results from experimental plots showed that coarsening of vegetation patterns lead to significantly higher erosion rates due to concentration of overland flow and higher flow velocities (Puigdefabregas, 2005; Bautista et al., 2007). When runoff becomes concentrated downslope it can lead to gully formation and these gullies are effective links for transferring water and sediment from hillslopes to valley bottoms and channels and consequently increase the hydrological connectivity (Poesen et al., 2003). Also human-made structures such as ditches, agricultural terraces and roads influence the hydrological connectivity (Croke and Mockler, 2001). At catchment and watershed scale the geomorphology of a landscape and human impacts in the floodplain, e.g. check dams and reservoirs, determine the hydrological connectivity. Especially sediment conveyance is impeded by fluvial barriers, buffers and discontinuities between landscape compartments (Fryirs et al., 2007).

The objective of our study was to model runoff and erosion for a semi-arid catchment using a multi-scale approach based on hydrological connectivity. Runoff and erosion have been simulated at the catchment scale but plot and hillslope scale features that influence the hydrological connectivity were taken into account. First we identified the relevant sinks of runoff and sediment at plot and hillslope scale and quantified their influence. Afterwards we incorporated the effects of these sinks in the LAPSUS runoff and erosion model by adjusting the infiltration capacity. Finally, runoff and erosion were simulated for four different scenarios to evaluate the influence of the identified sinks on the hydrological connectivity at the catchment scale.

6.2. Regional setting

We selected the Carcavo basin in Southeast Spain as our study area because of its representativeness for semi-arid catchments that are susceptible to erosion. The catchment is located in the province of Murcia, near the town of Cieza (38°13′ N; 1°31′ W). It is a small catchment of 30 km^2 with altitudes ranging from 220 to 850 metre. The area has an average annual rainfall of 300 mm and a potential evapotranspiration of 900 mm. Rainfall is bimodal with peaks in April and October and especially in autumn torrential rainstorms occur. The geology and geomorphology of the area consists of steep Jurassic carbonate mountains with calcareous pediments, hilly terrain with Keuper gypsum marls and Cretaceous and Miocene marls in the lower parts of the basin. Soils in the Carcavo basin are closely linked to the geology, with thin soils (Leptosols) on the limestone and dolomite mountains, weakly developed soils on marls (Regosols), other soils in which often (petro)calcic horizons have developed (Calcisols) on the stable pediments, and gypsiferious soils (Gypsisols) on the Keuper marls. Current land use in the study area consists of agricultural land (barley, olives, almonds and vineyards), abandoned land, reforested land and shrubland. In the 1970s large parts of the area were planted with pine (*Pinus halepensis*) for reforestation and soil conservation purposes. During the last decades part of the non-irrigated agriculture has been abandoned and is currently under different stages of secondary succession. For the simulation of runoff and erosion we selected the upper part of the Carcavo catchment, which comprised 498 ha (Figure 6.1). This part of the basin is characterised by the steep slopes and footslopes of the Sierra del Oro, which are partly reforested, and by a less dissected part with agricultural and abandoned land.

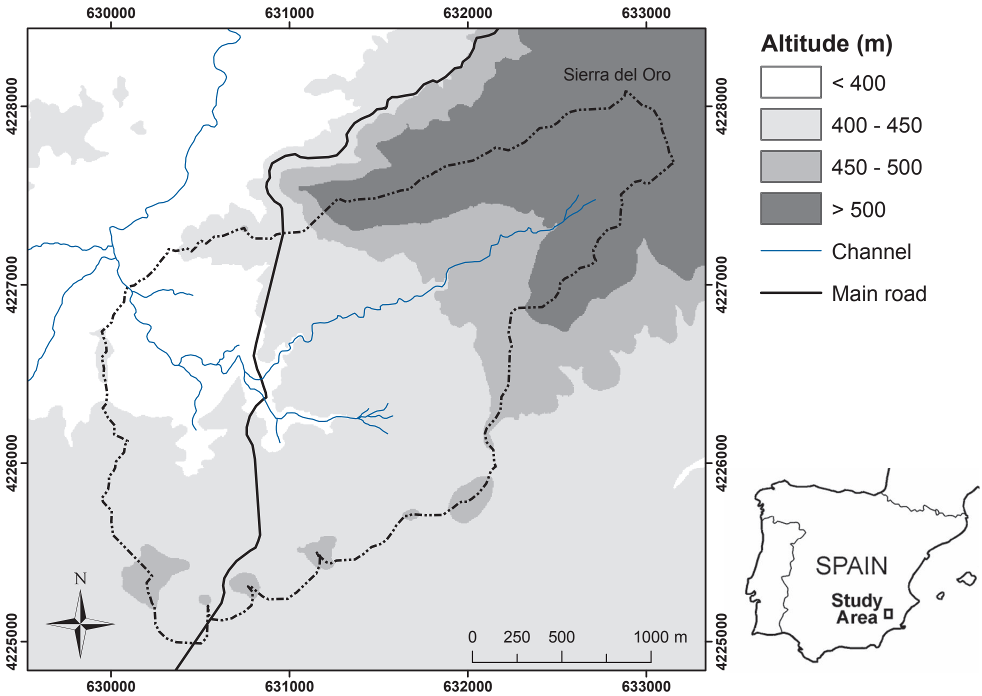


Figure 6.1. Location and topography of the upper part of the Carcavo basin

We focused on the two most relevant sinks for runoff and sediment within the catchment, which are vegetation patches at the plot scale and agricultural terraces at the hillslope scale. Vegetation in semi-arid areas is characterised by a heterogeneous pattern of bare soil and vegetated patches, which influences the hydrological connectivity and determines whether runoff is generated at the plot scale (Ludwig et al., 2005). At broader scales the spatial configuration of vegetation determines whether runoff generated at the plot scale will become connected and reach the channel (Puigdefabregas, 2005). However, in the agricultural part of the catchment the agricultural terraces have a major impact on the hydrological connectivity, since well functioning terraces can be large sinks for runoff and sediment. These agricultural terraces and earth dams are important features of traditional agricultural systems in hilly areas of Mediterranean countries. In the Carcavo basin about 39 percent of the agricultural land is currently terraced. Terraces have been constructed on hillslopes and in dry streambeds, while earth dams are mainly found in valley bottoms of undulating cereal fields.

6.3. Methodology

6.3.1. LAPSUS model

To simulate runoff and sediment dynamics we used the LAPSUS model (Schoorl and Veldkamp, 2001; Schoorl et al., 2002). LAPSUS is a dynamic landscape evolution model that can simulate erosion and sedimentation, based on a limited amount of input parameters. The model is based on two fundamental assumptions: (1) the potential energy of overland flow is the driving force for sediment transport (Kirkby, 1986), and (2) the difference between sediment input and output of a grid cell is equal to the net increase in storage, the continuity equation for sediment movement (Foster and Meyer, 1975). The model evaluates the rate of sediment transport by calculating the transport capacity of water flowing downslope from one grid cell to another as a function of discharge and slope gradient. A surplus of capacity is compensated by detachment of sediment, depending on the erodibility of the surface, which provokes erosion. When the rate of sediment in transport exceeds the local capacity, the surplus is deposited based on a sedimentability factor. The routing of runoff and resulting model calculations are based on a multiple flow algorithm, which allows for a better representation of divergent and convergent properties of topography (Holmgren, 1994). The model can be used for both short and long term applications by varying the time step. For our study we simulated runoff and erosion based on 1-hour time steps for a six day rainfall period in November 2006.

The main input data for the LAPSUS model are a digital elevation model (DEM), rainfall data, infiltration data, erodibility characteristics and soil depth. To represent the topography we used a 5 metre resolution DEM, which was derived from a 1:5000 topographic map with 5 metre contour lines. Although this resolution is relatively high, it still did not represent well enough the agricultural terraces, since these generally have height difference of only one to two metres. Therefore we had to adapt the DEM to include the topography of the terraces. First we digitised the terraces in ArcGIS 9.0 (ESRI, Redlands, US) and calculated the average altitude for these polygons using zonal statistics. Next, the polygons were converted to raster with mean altitude as value, and this map was subtracted from the DEM. The difference in altitude was multiplied with a factor 0.75, to include some effect of the original topography, because most terraces are not completely flat, especially at the sides. Finally, the calculated altitude difference was added to the DEM, to represent a more realistic terrace topography.

For rainfall we used data from a large event in November 2006, which was recorded with a tipping bucket rain gauge in the centre of the catchment. The event included several rainstorms within six days with a total rainfall of 157 mm, which was half of the average annual rainfall. Although most rainfall was of low intensity (Figure 6.2), the rainstorm of the last day had a high intensity with a maximum of 50 mm h^{-1}, based on a 10-minute interval. The recurrence time of this last rainstorm was about two years, based on a 36-year rainfall series of the Almadenes weather station. However, when rainfall of the previous days was taken into account the recurrence time for the 10-day rainfall sum was about 23 years. Although the spatial distribution of rainfall can be highly variable (González-Hidalgo et al., 2001), we considered the catchment to be small enough to neglect the spatial variability of rainfall.

The most important input for our modelling exercise is infiltration. Modification of the infiltration capacity enables the incorporation of runoff sinks, which finally determine whether runoff will occur. In the original version of the LAPSUS model a fixed value for infiltration is used for each soil type and land use (Schoorl et al., 2002). Buis and Veldkamp (2008) modified the infiltration parameters to better represent the spatial diversity in water redistribution and availability within arid catchments. We further elaborated the infiltration module by making it dynamic and dependent on measured parameters. Runoff threshold and runoff coefficient were selected to describe the infiltration characteristics in the LAPSUS model. These parameters were derived from a database of rainfall simulations, which is described in Section 6.3.2. The model first calculates the amount of rainfall available for runoff after subtracting the runoff threshold. This value is then multiplied with the runoff coefficient, which determines the amount of runoff for each grid cell. Next, the model determines the routing of runoff, based on the multiple flow algorithm, and evaluates whether infiltration capacity is still available. When runoff has been determined for the entire catchment, the model calculates the transport capacity for sediment, which determines whether sediment is eroded or deposited. We estimated that 15% of the runoff threshold and 10% of the storage and infiltration capacity of the terraces is recovered per day due to evapotranspiration and drainage to deeper soil horizons.

The erodibility factor and soil depth were estimated for each substrate (Table 6.1). The distribution of the substrates was derived from a 1:50,000 geological map and an additional field survey, which was required in order to map the location of calcretes. The erodibility factor determines how easy sediment is detached with low values indicating a low erodibility. For all agricultural land we added a value of 0.002 to the erodibility factor to take account of ploughing, which increases the erodibility. The last input variable is a sedimentation factor, which determines how easy sediment can be deposited. For this factor we distinguished between bare soil and vegetation, since vegetation increases the surface roughness and promotes sedimentation (Bochet et al., 2002). The sedimentability factor was set a factor 5 higher for vegetated areas. All spatial input data were converted to raster files with a resolution of 5 metre.

Table 6.1. Occurrence of substrates within the simulated catchment and the estimated parameters

Substrate	Occurrence (%)	Erodibility factor (m^{-1})	Soil depth (m)
Slope deposits	27.4	0.0025	0.40
Slope deposits with calcrete	13.6	0.0015	0.25
Marl	39.4	0.0050	0.40
Keuper	9.3	0.0035	0.20
Limestone / dolomite	10.3	0.0005	0.15

6.3.2. Influence of vegetation patches

Vegetation in semi-arid environments is characterised by heterogeneous patterns of bare and vegetated patches. The bare patches are generally formed by bare rock and crusted soils with poor soil structure and low infiltration rates, whereas vegetated patches have better soil properties, higher organic matter content and a better aggregation, which results in a higher infiltration capacity. This makes overland flow highly discontinuous with bare patches as runoff generating areas and vegetated patches as runoff sinks (Bergkamp, 1998; Cammeraat and Imeson, 1999). Soil moisture measurements in the study area confirmed the higher infiltration capacity under vegetation (Figure 6.2). FDR (Frequency Domain Reflectometry) sensors at 5 cm depth recorded soil moisture at a bare and a vegetated patch. Soil moisture under vegetation was on average 7 volume percent higher, and reacted more quickly to rainfall, which indicates better infiltration capacity (Bergkamp et al., 1998).

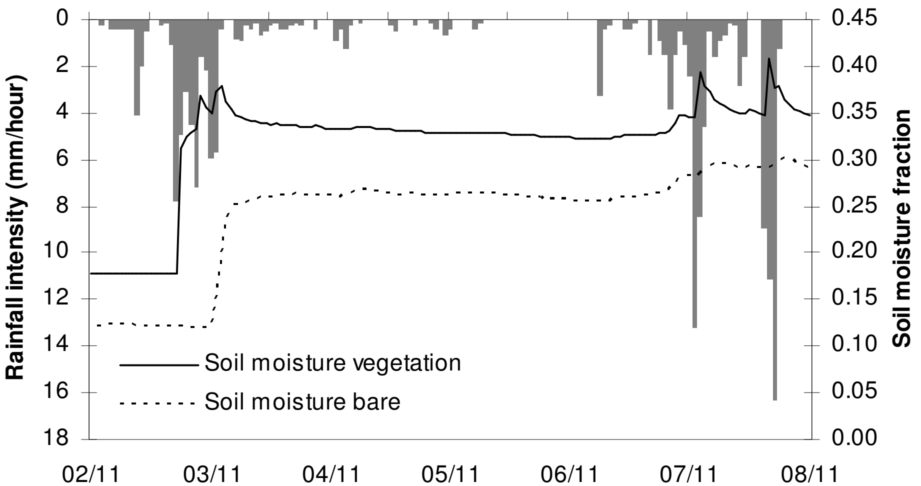


Figure 6.2. Measured soil moisture under bare soil and vegetation on marl during a six day rainfall period in November 2006

To simulate the influence of vegetation patterns on runoff and erosion we differentiated the infiltration parameters for bare and vegetated areas. These parameters, runoff threshold and runoff coefficient, were derived from a database of rainfall simulation experiments. The database comprised data from studies that were applied in similar semi-arid areas, most of them being in Southeast Spain: Boix Fayos et al. (1995), Calvo-Cases et al. (2003), Cammeraat and Imeson (1999), Cammeraat et al. (2002), Castro et al. (2006), Ceballos et al. (2002), Cerda (1997a), Cerda (1997b), Cerda (1998), Cerda (2001), Coppus and Pulleman (1996), Imeson et al. (1998), Lasanta et al. (2000), Meerkerk et al. (submitted1), Prinsen (unpublished), Quinton et al. (1997) and Seeger (2007). We only selected data from experiments that were applied on substrates and land uses that occur within the study area, which resulted in 285 observations. Although these experiments were used to study a wide range of research questions and not all studies measured or described the same variables, the size of the database is large enough to obtain general and statistically significant relationships between environmental factors and measured parameters. Runoff threshold and runoff coefficient were selected to describe the infiltration characteristics in the LAPSUS model. Runoff threshold was defined as the amount of rainfall until runoff started, which was often derived from the observed time to runoff. Runoff coefficient is the ratio of observed runoff to applied rainfall. However, combining both parameters will overestimate the total infiltration capacity, since part of the rainfall was already included in the runoff threshold. Hence, the runoff coefficient was recalculated based on the amount of rainfall minus the runoff threshold. Next, we classified all data for the different combinations of land use (shrubland, abandoned land, and agricultural land), substrate (marl, slope deposits and limestone) and vegetation cover (bare and vegetated). For these classes we calculated the mean values for runoff threshold (n=143) and runoff coefficient (n=218).

Upscaling of the vegetation patterns was required to include the influence of vegetation patches for the entire catchment. Imeson and Prinsen (2004) and Lesschen et al. (2008a) found a linear relationship between vegetation cover and spatial metrics that described the vegetation patterns. This enabled us to use vegetation cover as a proxy for spatial vegetation structures in landscapes with spotted vegetation patterns. Lesschen et al. (submitted) created a fractional vegetation cover map for the Carcavo basin based on a linear regression of vegetation cover, as derived from detailed aerial photographs, and the green and red reflectances from a QuickBird image (R^2 of 0.91). Since the rainfall simulation database contained insufficient data about fractional vegetation cover we could only differentiate two classes of vegetation cover, i.e. bare and vegetated areas. The threshold between these two classes was set at 30 percent, since several studies showed that runoff and erosion increase drastically below 30 percent vegetation cover (Francis and Thornes, 1990; Quinton et al., 1997; Ludwig et al., 2002).

6.3.3. Influence of agricultural terraces

Agricultural terraces are important sinks for runoff and erosion at the hillslope scale. These terraces were originally constructed to reduce soil erosion and to intercept runoff by decreasing the general slope (Morgan, 1995). However, the effects of terracing are not only positive, since the area near the terrace rim is now under influence of a steep hydraulic gradient, which may lead to gully erosion and piping (Faulkner et al., 2003). To include the influence of the terraces in LAPSUS we first digitised the terrace walls in ArcGIS, based on a QuickBird satellite image of 2006. During a field survey we checked which terraces were intact and which were collapsed (Figure 6.3). In total about 500 terrace walls were identified, of which 127 were collapsed. Most of the failed terraces were located on abandoned fields, which are more vulnerable to failure due to the lack of maintenance and increased runoff (Lesschen et al., 2008b). The terrace wall map was converted to raster and intact terraces were assigned an additional infiltration capacity. Although in reality this additional storage and infiltration capacity depends on the terrace characteristics, e.g. size and slope of the terrace and height of the terrace rim, no such data were available to create a variable storage capacity for the terraces. Instead we used a single value of 50 m^3 of water that can be additionally stored and infiltrated at each terrace.

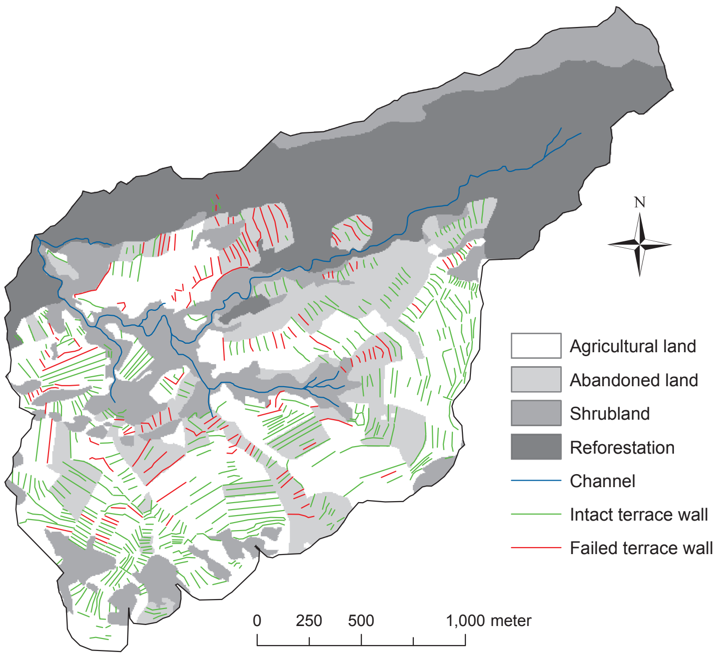


Figure 6.3. Land use of the simulated catchment and locations of intact and failed terrace walls on (abandoned) agricultural land

6.3.4. Scenarios

To evaluate the influence of the identified sinks on the hydrological connectivity at catchment scale we used four different scenarios for the simulation of runoff and erosion. Besides a scenario that represents the current situation, two other scenarios were used to assess the influence of vegetation and agricultural terraces respectively and a fourth scenario was used to simulate the influence of a soil and water conservation practice in which all failed terraces are repaired. Accordingly the four scenarios are:

1. Current situation
2. No vegetation patterns
3. No terraces
4. All terraces function

The first scenario includes the influence of vegetation patterns at plot scale as well as the influence of intact and collapsed terraces at hillslope scale. For the second scenario we adapted the values of the runoff threshold and runoff coefficient to exclude the influence of vegetation patterns. Based on the rainfall simulation database we calculated new values for each combination of land use and substrate without differentiating between bare and vegetated areas. For the third scenario we did not include the extra infiltration capacity for terraces and used the original DEM without the detailed terrace representation. Finally, for the last scenario we modified the input file with additional infiltration capacity and gave the failed terraces the same infiltration capacity as the other terraces.

6.3.5. Verification of the model results

To verify the spatial distribution of the model results we compared the simulated runoff map with observed concentrated flow paths. These connectivity patterns were mapped after the rainfall event of November 2006 (Meerkerk et al., submitted2). Since the soil was already saturated from the rainfall of the previous days the intense rainfall on 8 November resulted in significant amounts of erosion, as observed by completely filled sediment collectors, rills on agricultural fields, and failure of some agricultural terraces. Within the agricultural part of the sub-catchment all signs of concentrated flow were recorded with a handheld GPS. These flow paths were visible as rills and gullies, but also as areas with clear signs of concentrated overland flow as observed by the removal of small branches, leaves and other litter material. Although concentrated flow was also observed on roads, we excluded these flow paths since the influence of roads is not included in the model. In addition to the mapped connectivity patterns on agricultural land, we added the flow paths in the channels.

6.4. Results

Based on the analysis of rainfall simulation data we calculated the mean runoff threshold and runoff coefficient for the different land use - substrate - vegetation cover classes (Figure 6.4). The runoff threshold for bare conditions was lower compared to vegetated conditions. For bare areas runoff starts approximately after 7 mm of rainfall, while under vegetation the runoff threshold is about 13 mm. The results also showed that the threshold on abandoned fields is much lower than for shrubland and agricultural land. The runoff coefficient also clearly showed the difference between bare and vegetated areas, on average the runoff coefficient was 0.19 higher under bare conditions. The runoff coefficient was also higher for abandoned land compared to shrubland for both vegetated and bare conditions.

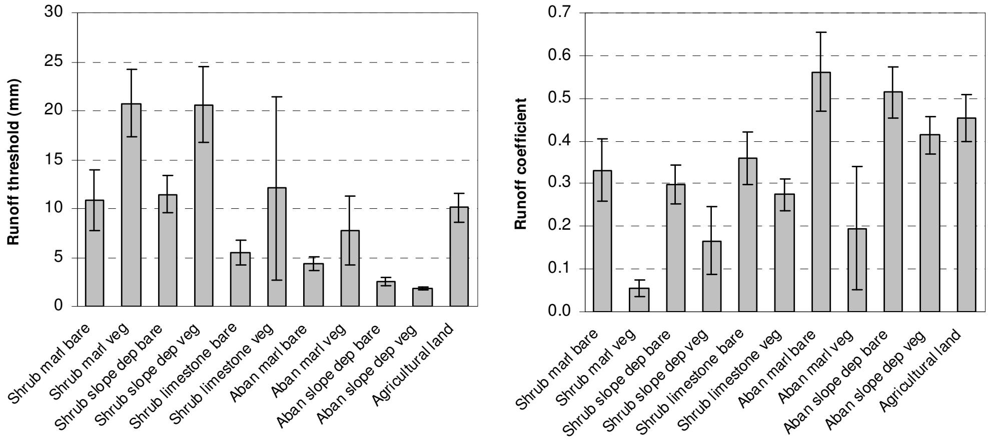


Figure 6.4. Mean runoff threshold and runoff coefficient (RC) for the different classes of land use, substrate and vegetation cover (error bars indicate standard error of the mean)

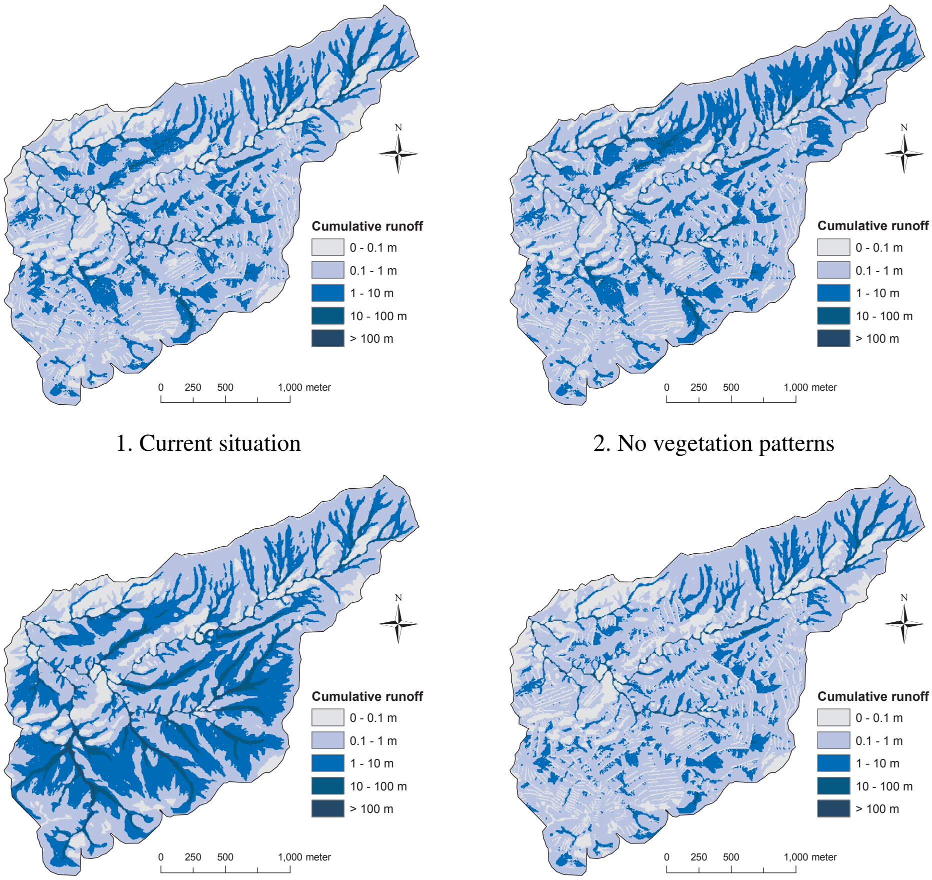
The results of the runoff and erosion simulations with the LAPSUS model are summarised in Table 6.2 for the four scenarios. Total runoff is the sum of discharge at the outlet of the catchment. RC is the runoff coefficient for the entire catchment, i.e. total runoff divided by the total rainfall. Erosion and sedimentation are the sum of erosion and sedimentation respectively for each time step. Sediment yield is the sum of the transported sediment at the catchment outlet and is consequently the net erosion for the entire catchment. Finally, SDR is the sediment delivery ratio, which is the fraction of erosion that is transported out of the catchment.

For the current situation (Scenario 1) a runoff coefficient of 0.068 and a sediment yield of 2.5 Mg ha^{-1} were predicted. The sediment delivery ratio was only 0.063, which means that most of the sediment is deposited within the catchment. For the second scenario without the influence of vegetation patterns more runoff was produced and the sediment yield was a factor 3 higher compared to current situation. In the third scenario without agricultural terraces the runoff coefficient and sediment yield were much higher, while erosion and sedimentation remained relatively low, which means that more sediment was transported out of the catchment, as indicated by the high sediment delivery ratio. In the last scenario with all terraces intact less runoff was produced and erosion and sedimentation were lower. However, sediment yield and sediment delivery ratio were somewhat higher than in the scenario for the current situation.

Table 6.2. Summary of model outputs for the simulated scenarios

Scenario	**Total runoff** (1000 m^3)	**RC**	**Erosion** (Mg ha^{-1})	**Sedimentation** (Mg ha^{-1})	**Sediment yield** (Mg ha^{-1})	**SDR**
1 Current situation	52.9	0.068	38.1	35.7	2.5	0.063
2 No vegetation patterns	76.5	0.098	56.4	49.1	7.4	0.129
3 No terraces	203	0.259	84.8	61.7	23.4	0.272
4 All terraces intact	40.4	0.052	32.0	29.1	3.0	0.091

Figure 6.5 shows the results of the cumulative runoff for the four scenarios. The cumulative runoff is the sum of runoff for all time steps that passed a grid cell and is expressed in meter height. In the first scenario, representing the current situation, runoff from the Sierra del Oro was well connected to the main channel, while runoff from the agricultural fields was mainly retained by the agricultural terraces. Only fields close to the channel were connected and contributed to the discharge in the channel. For the second scenario without the influence of vegetation patterns, the runoff intensity was higher and especially the slopes of the Sierra del Oro produced more runoff. For the southern part of the catchment somewhat more runoff was produced, but this did not increase the hydrological connectivity. The third scenario without the influence of agricultural terraces showed a very strong increase in runoff in the southern part of the catchment where runoff of all hillslopes became connected to the channel. Whereas runoff from the hillslopes in the last scenario with all terraces intact was not connected to the channel in the agricultural part of the catchment. All locally produced runoff was retained by terraces downslope and the channel was only supplied by runoff from nearby areas.



1. Current situation 2. No vegetation patterns

3. No terraces 4. All terraces function

Figure 6.5. Cumulative runoff for the four scenarios

The results of the simulation of erosion and sedimentation resulted in different magnitudes of sediment dynamics, but also the patterns of erosion and sedimentation were different for the four scenarios (Figure 6.6). In the current situation most erosion occurred on the steep slopes of the Sierra del Oro in the northern part of the catchment and on areas near the channel, while sedimentation occurred on the footslopes and on agricultural terraces. In the channel an alternation of areas with erosion and areas with sedimentation was observed. For the second scenario without the influence of vegetation patterns the erosion and sedimentation patterns were similar, but the amount of erosion and sedimentation was higher. The third scenario without any terrace influences resulted in a completely different pattern in the agricultural part of the catchment. High erosion rates occurred in all valley bottoms, while sedimentation was limited to the footslopes of the Sierra del Oro and the channels. Finally, the fourth scenario showed the positive effects of well-maintained terraces, which retained sediment and prevented erosion in the agricultural part of the catchment.

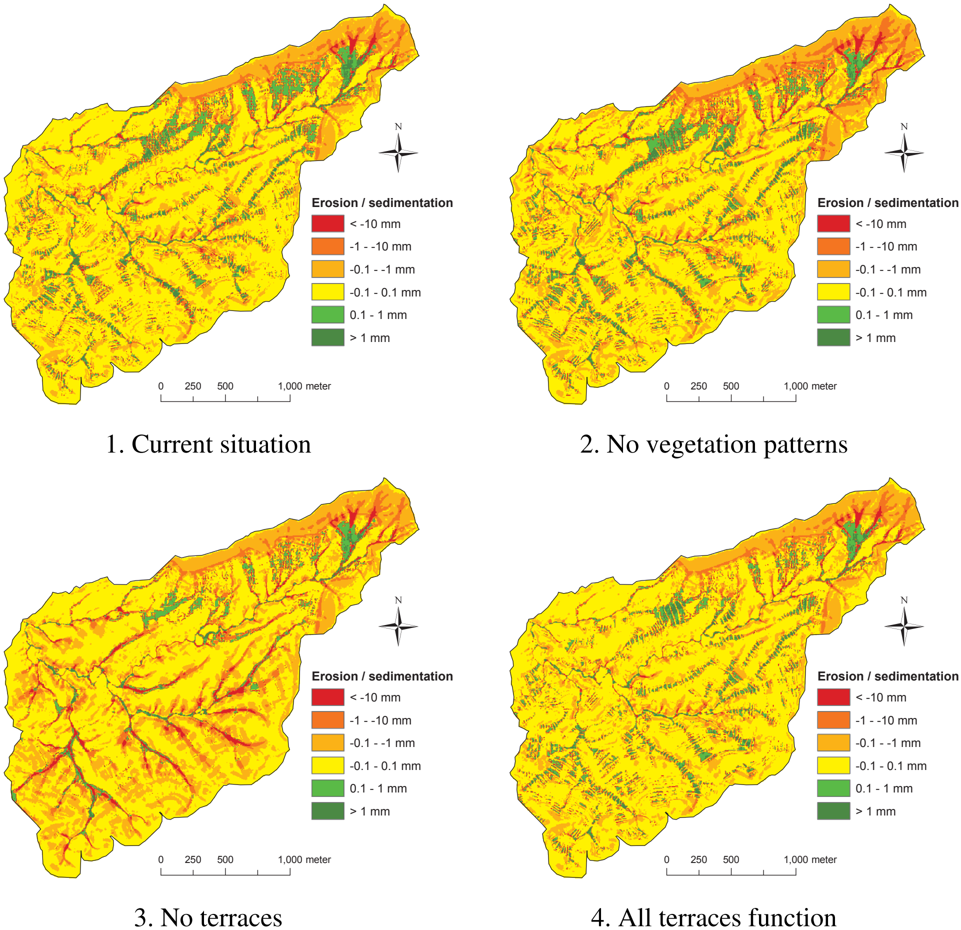


Figure 6.6. Erosion (negative values) and sedimentation (positive values) patterns for the four scenarios

6.5. Discussion

The results of our runoff and erosion simulations showed that vegetation patterns and agricultural terraces have a major influence on the connectivity of runoff and sediment. Comparing the first and second scenario, excluding the influence of vegetation patterns led to an overestimation of runoff and erosion, especially in areas with a higher vegetation cover. However, the patterns of hydrological connectivity at catchment scale did not change. In contrast, the scenario without the influence of agricultural terraces showed a major impact on the hydrological connectivity at catchment scale. For this scenario total runoff was four times higher and sediment yield almost ten times higher, due to lowering of the runoff threshold on agricultural fields (Cammeraat, 2004). Consequently all runoff from the hillslopes of the entire catchment was connected to the channel. Nevertheless the last scenario showed the important function of agricultural terraces for soil and water conservation. In this case most of the runoff and sediment from the agricultural part of the catchment was not connected to the channel and retained by the agricultural terraces. Although total runoff and gross erosion were lower, the sediment yield for this scenario was somewhat higher. This might be explained by a lower sediment delivery from the agricultural land to the channel, which can increase the erosive force of the water in the channel and lead to more erosion in the channel and to a higher sediment transport out of the catchment. This effect is similar to the ‘clear water effect’ behind checkdams (Castillo et al., 2007).

Although erosion on agricultural land is generally higher compared to semi-natural areas (Kosmas et al., 1997; Cammeraat, 2004), the hydrological connectivity was lower, due to the influence of agricultural terraces and other water conservation structures. As a result the sediment yield from agricultural areas is limited to fields near the channel and most of the sediment is retained on agricultural fields. However, during extreme events the accumulated sediment might still be transported to the channel due to failure of terraces, which will increase the hydrological connectivity. Furthermore, in case of agricultural abandonment part of the terraces might collapse, which will increase the hydrological connectivity. This was observed in several parts of the catchment, where gullies through terrace walls increased the connectivity.

To verify the spatial distribution of the simulation, we compared the modelled runoff map of scenario 1 with the observed concentrated flow (Figure 6.7). This map showed that the overall runoff pattern was fairly well simulated with low connectivity between the hillslopes and the channel in the southern part of the catchment. Field observations confirmed that flow inside the channel was fully connected, as was simulated by the model. Furthermore, the highest cumulative runoff class (>100 m) coincided very well with the location of the actual channels. However, not all individual flow paths were correctly simulated, probably due to lack of detailed input data. A more detailed DEM, for instance derived by LIDAR, is necessary to capture all topographical features such as small ditches, agricultural terraces and field boundaries, which do determine local surface runoff. Also roads are important pathways for runoff, which frequently lead to gully initiation due to runoff concentration (Croke and Mockler, 2001; Meerkerk et al., submitted2). Furthermore, identification of concentrated flow in the field is not always straightforward, since not all concentrated flow is visible as rills. On freshly ploughed fields the development of rills is enhanced due to the high erodibility. However, on abandoned land the formation of rills is reduced by the presence of soil crusts and most of the overland flow will not leave clear traces. This makes identification of flow paths in the field difficult and might lead to mismatches between simulated and observed connectivity patterns.

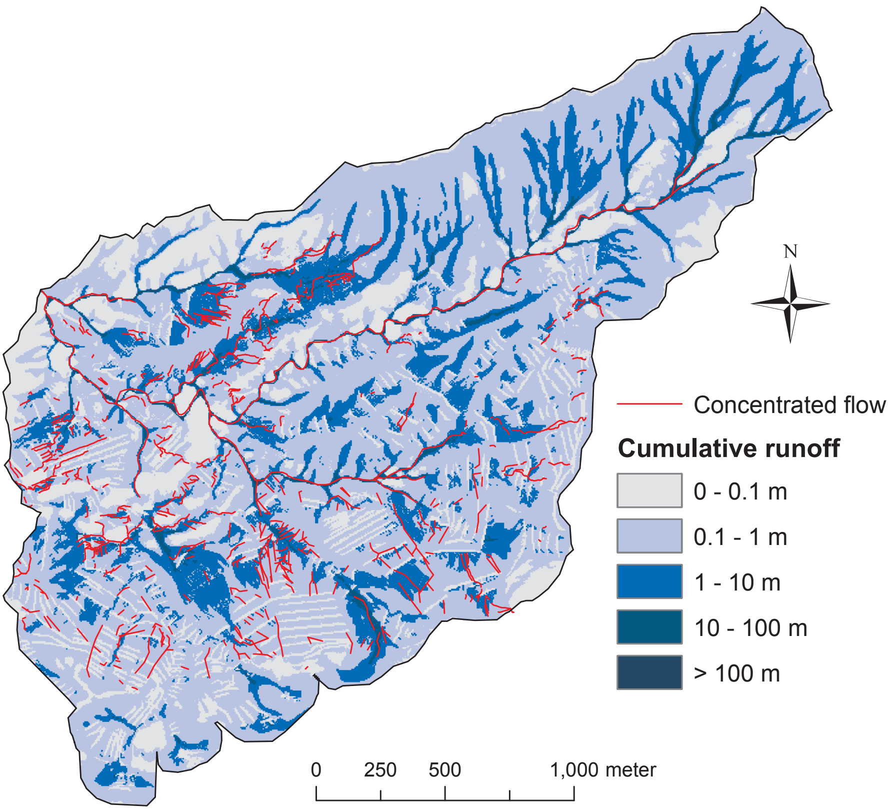


Figure 6.7. Overlay of observed concentrated flow (red), based on Meerkerk et al. (submitted2) and simulated runoff (blue) for scenario 1 (current situation)

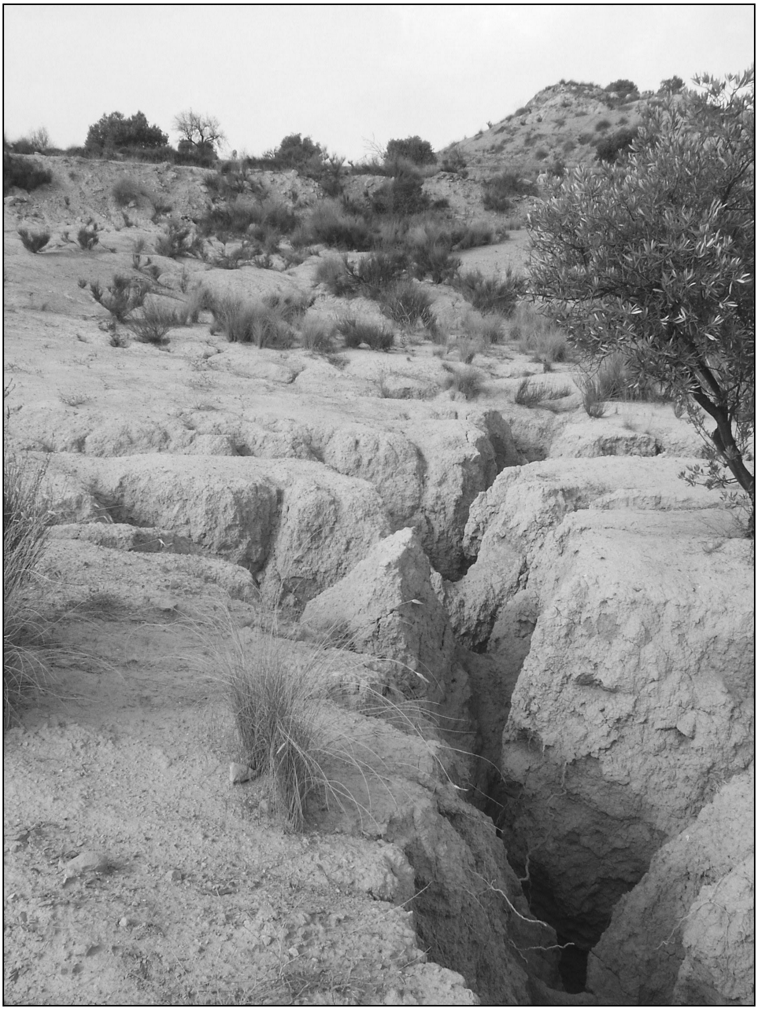
Besides the verification of spatial runoff patterns, also the simulated total runoff and sediment yield at the outlet of the catchment has to be assessed, since these determine the impact on off-site effects such as flash flooding risk and reservoir sedimentation. Unfortunately, no monitoring data were available to validate total discharge and sediment yield for the modelled catchment. However, based on other studies for comparable semi-arid catchments, we were able to assess whether the simulated runoff and sediment yield were realistic. The simulated runoff coefficient for the entire catchment was 0.068, based on the current situation scenario. Although this value is quite low, it seems a reasonable value considering the low intensity of most of the rainfall and the presence of agricultural terraces in the southern part of the catchment, which are major sinks for runoff. Other studies also found rather low runoff coefficients for semi-arid catchments. Martín-Vide et al. (1999) reported runoff coefficients ranging between 0.04 and 0.15 for a 30 km^2 catchment in Northeast Spain and Michaud and Sorooshian (1994) and Ye et al. (1997) found runoff coefficients around 0.1 for respectively the semi-arid Walnut Gulch catchment in New Mexico and ephemeral catchments in Western Australia. For sediment yield less comparable studies were available. De Vente and Poesen (2005) demonstrated the relation between drainage area and area-specific sediment yield based on Spanish data, which shows first an increase in area specific sediment yield and afterwards a decrease for drainage areas above 100 km^2. However, no data were available for catchments with drainage areas between 0.1 and 30 km^2, which have the highest area-specific sediment yields. Nevertheless, a maximum area-specific sediment yield of 20-40 Mg ha^{-1} year^{-1} for small scale catchments (1-10 km^2) can be inferred from their graphs. This is higher than our simulated sediment yield of 2.5 Mg ha^{-1}, but as discussed before, rainfall intensity was low for most of the event, while in semi-arid landscapes most erosion can be contributed to extreme events (Cammeraat, 2004; Boardman, 2006).

Modelling runoff and erosion at catchment scale makes simplifications of hydrological processes and sediment dynamics unavoidable. Runoff generation, which is the most important process for our modelling approach, was simplified by using a runoff threshold and runoff coefficient, while runoff generation in reality is controlled by complex and dynamic interactions between rainfall characteristics and soil and surface properties (Martinez-Mena et al., 1998). More sophisticated approaches such as the Green-Ampt equation require too much input data at the catchment scale. The values for runoff threshold and runoff coefficient were mean values from all rainfall simulation experiments, for which we did not make a distinction between dry and wet preceding conditions. However, Cammeraat and Imeson (1999) and Cammeraat (2004) showed that runoff curves can differ greatly depending on the initial soil moisture conditions. Nevertheless, in our modelling approach the infiltration capacity has been made dynamically and rainfall from the previous time steps is taken into account. Another simplification was the non-variable infiltration and storage capacity for the agricultural terraces, which lead to underestimation of runoff on small terraces, while large terraces with well constructed rims could have retained more runoff. Nevertheless, the overall runoff pattern appeared to be realistic, since field observations showed that only limited areas were connected to the channel and most agricultural hillslopes retained all runoff. When a more detailed DEM is available the sink function of agricultural terraces can also be simulated dynamically based on the algorithm of Temme et al. (2006), which models the sediment buffer function of depressions. A final simplification is the exclusion of bench terraces from the reforested areas in the simulation, which might have led to an overestimation of runoff and erosion in the northern part of the catchment. However, Maestre and Cortina (2004) and Hooke (2006) have questioned the effectiveness of these terraces and De Wit and Brouwer (1998) found higher runoff and sediment yields in reforested areas due to gullying through the bench terraces.

Modelling runoff and erosion without taking into account the relevant sinks from the plot and hillslope scale will lead to overestimation of runoff and sediment yield as we demonstrated in our study. Otherwise, models can predict acceptable soil loss and discharge at the outlet, but with incorrect patterns of source and sink areas (Jetten et al., 2003). Application of erosion models for soil and water conservation purposes requires correct simulation of runoff and erosion patterns in order to identify the hotspots where mitigation measures should be applied. Although Jetten et al. (2003) were negative about the ability of catchment based models to determine optimal locations for water conservation measures, e.g. grass strips or terraces, our simulations gave satisfactory results for the implementation of a conservation practice, i.e. restoration of existing failed terraces. In the current situation (scenario 1) several areas with high runoff production in the agricultural part can be identified, restoration of the failed terraces at these sites (scenario 4) significantly reduced runoff and erosion. Also other measures could be evaluated, e.g. grass strips, by adapting infiltration and erodibility input data.

6.6. Conclusions

Our new approach is based on the identification of hydrological sinks which are integrated in a distributed runoff-erosion model. The results of the simulations show that the spatial distribution of sinks at plot and hillslope scale, in this case vegetation patches and agricultural terraces, determine for a large part the hydrological connectivity at catchment scale. Runoff and sediment dynamics are therefore non-linear and scale dependent and are to a large extent determined by the spatial distribution of hydrological sinks. Especially the additional infiltration and storage capacity of agricultural terraces is important to take into account, since exclusion will highly overestimate runoff and erosion in catchments with these terraces and erroneous runoff and erosion patterns will be simulated. Although LAPSUS is a relatively simple model we demonstrate that the model is able to reproduce the observed patterns of hydrological connectivity and to predict realistic values for total runoff and sediment yield. Distributed hydrological and erosion models should take account of relevant sinks at finer spatial scales in order to simulate patterns of runoff and erosion correctly.



7. Synthesis

This final chapter will summarize the conclusions from the previous chapters and discuss the main research questions of this thesis. In the following sections the results and implications of this research in relation to the three central themes, i.e. soil erosion, scale issues and agricultural land abandonment, will be synthesised. Finally, some recommendations and ideas for future research will be proposed.

7.1. Conclusions

Land use change modelling shows that future land abandonment will most likely occur on fields without irrigation possibilities and located on slopes or in valley bottoms in more remote areas. Gully erosion risk is higher on abandoned fields compared to cultivated fields. Especially fields located around channel heads or near channel walls are vulnerable to gully erosion. The combination of a higher gully erosion risk on abandoned fields and an expected increase of land abandonment will enhance the negative off-site effects of gully erosion. Nevertheless, identification of potentially vulnerable areas enables timely mitigation of gully erosion by applying preventive soil and water conservation measures.

Vegetation recovery and improvement of soil properties after land abandonment are slow under the semi-arid conditions of the Carcavo basin. Secondary vegetation succession on calcrete soils appears to be faster than on marl soils, probably because of the higher water availability due to higher rock fragment cover on calcrete soils. Under vegetated patches organic matter content, aggregate stability and electrical conductivity are significantly higher compared to those on bare patches. Most of the spatial metrics that describe vegetation patterns in landscapes with spotted vegetation have a linear relationship fractional vegetation cover, which offers the possibility for upscaling spotted vegetation patterns.

In the Carcavo basin more than half of the abandoned fields have moderate to severe erosion, especially in the case of terrace failure. Factors that increase the risk of terrace failure are land abandonment, steeper terrace slope, soils with loamy texture, valley bottom position and shrubs on the terrace wall. Terracing, although intended as conservation practice, often enhances erosion, especially after abandonment and in substrates susceptible to piping. The two main soil and water conservation measures to mitigate erosion on abandoned fields are maintenance of terrace walls and revegetation with indigenous grass species on zones with concentrated flow.

Vegetation patterns have a large influence on runoff and erosion in semi-arid environments. Upscaling of these patterns is therefore necessary to include their influence in runoff and erosion modelling at broader scales. From the ten evaluated vegetation indices a regression of the green and red reflectance, derived from a QuickBird image, had the best fit with observed vegetation cover from detailed aerial photographs. DVI based vegetation indices are less suitable for determination of vegetation cover in semi-arid areas because of the atypical spectral behaviour of drought tolerant vegetation.

The results of runoff and erosion modelling, based on a multi-scale approach, show that the spatial distribution of sinks at plot and hillslope scale, in this case vegetation patches and agricultural terraces, largely determine the hydrological connectivity at catchment scale. Runoff and sediment dynamics are therefore non-linear and scale dependent and to a large extent determined by the spatial distribution of hydrological sinks. Distributed hydrological and erosion models should therefore take account of relevant sinks at finer spatial scales in order to simulate patterns of runoff and erosion correctly at broader scales.

The above stated conclusions were the conclusions from the different chapters, which were used to answer the three main research questions of this thesis.

1. Where does agricultural land abandonment occur and how do vegetation and soil properties change after abandonment?

In the Carcavo basin land abandonment is frequently occurring and currently 17 percent of the total agricultural area has been abandoned. Future land abandonment will most likely occur on fields without irrigation possibilities located on slopes or in valley bottoms in more remote areas. After abandonment the secondary vegetation succession starts and vegetation cover, size of vegetation patches and connection between patches increase with time since abandonment. Organic carbon content and aggregate stability also increase with time since abandonment, but mainly under vegetated patches. As a consequence the spatial heterogeneity in vegetation and soil properties increases, which makes overland flow highly discontinuous. However, these changes are slow under the semi-arid conditions and it takes at least 40 years before vegetation and soil are similar to the semi-natural vegetation.

2. Which are the main soil erosion processes on abandoned land and how to mitigate them?

Land abandonment in semi-arid environments frequently enhances soil erosion during the first years. The absence of ploughing and slow vegetation recovery cause the formation of soil crusts with low infiltration rates, resulting in increased runoff and gully erosion risk.

Especially failure of agricultural terraces due to gully erosion and piping causes high sediment losses. Potential soil and water conservation practices to mitigate soil erosion after abandonment are the maintenance of terrace walls or revegetation with indigenous grass species on spots with concentrated flow. These mitigation measures should be aimed at erosion hotspots where intervention will have the largest reduction of the hydrological connectivity, e.g. on failed terraces in valley bottoms.

3. How to integrate plot and hillslope scale influences in runoff and erosion modelling at catchment scale?

Runoff and sediment dynamics are non-linear, scale dependent and largely determined by the spatial distribution of hydrological sinks. The concept of hydrological connectivity is scale independent which makes it suitable to take account of scale dependency by identifying sources and sinks of runoff and sediment and their linkages at different spatial scales. In the Carcavo basin the relevant hydrological sinks are vegetation patches at plot scale and agricultural terraces at hillslope scale. Their influences can be quantified and GIS and remote sensing techniques can be used for upscaling to the catchment scale, as was shown for vegetation cover in Chapter 5. Finally, these results can be integrated in a distributed model, such as LAPSUS, to simulate patterns of runoff and erosion.

7.2. Soil erosion

Three key aspects of this thesis can be distinguished that contribute to the further understanding, improved predictions and mitigation of soil erosion. These are the study to the causes of soil erosion and especially terrace failure on abandoned land, the application of hydrological connectivity for runoff and erosion modelling, and the practical aspect about how and where to apply soil erosion mitigation measures. This thesis demonstrated that abandoned fields in semi-arid environments are vulnerable to soil erosion. Especially gully erosion through terrace walls is frequently occurring and can lead to high erosion rates. Numerous studies show that gully erosion is an important source of sediment (Poesen et al., 2003). For Spain gully erosion attributes about 50 to 80 percent to total sediment yield (Poesen et al., 1996; Casali et al., 2000; Poesen et al., 2002). In addition, these gullies increase the hydrological connectivity between the hillslope and the channel, which will enhance peak flows and the risk on flash floods. The results of Chapter 2 showed that especially fields located around channel heads or near channel walls are vulnerable to gully erosion due to land abandonment, which means that the connectivity between the hillslope and channel will increase. Mitigation of erosion should therefore focus on hotspot areas between the channel and the hillslope. Here simple mitigation measures can significantly decrease the hydrological connectivity and sediment losses.

Hydrological connectivity has shown to be a useful concept for determination of runoff generating areas at different scales. In hydrological and geomorphological research the concept is only recently being applied (Bracken and Croke, 2007). However, in ecology connectivity is a frequently concept to study spatially structured populations (e.g. Metzger and Decamps, 1997). The concept is a way forward to the understanding of hillslope and catchment responses to rainfall and provides a basis for integrating processes of runoff generation and landscape characteristics at different scales. Connectivity mapping in the field is a practical and relatively quick method to assess spatial patterns of runoff and erosion after an event. The concept of hydrological connectivity is also very useful for the definition of source and sink areas of runoff and sediment and their linkages at different scales, which is crucial for the mitigation of the off-site effects of soil erosion. Such mitigation measures should aim at erosion hotspots where intervention will have the largest reduction of the hydrological connectivity.

Vegetation is one of the key factors that control soil erosion as was concluded in this thesis and by many other studies (e.g. Thornes, 1990; Quinton et al., 1997; Cerdà, 1998). At plot scale the vegetation canopy, roots and improvement of soil properties influence soil erosion. At broader scales the spatial distribution of vegetation becomes important, since runoff and erosion processes are largely determined by the spatial configuration of vegetation patches (Bergkamp et al., 1996; Cammeraat, 2004; Ludwig et al., 2005). Even at catchment scale vegetation patterns had a considerable influence on runoff and erosion rates as turned out from the simulations with the LAPSUS model. However, under semi-arid conditions, as in the Carcavo basin, a high vegetation cover will not occur. These semi-natural shrublands typically have a vegetation cover of about 50%, which is sufficient to control runoff and erosion. Soil erosion rates in semi-natural areas in Spain are therefore generally low, whereas the highest rates are often found in agricultural systems (Cammeraat, 2004; Martínez-Fernández and Esteve, 2005). However, in the perception of farmers soil erosion is often not considered to be a major issue (Oñate and Peco, 2005), since not soil fertility but water availability is often the limiting factor in these semi-arid environments. Mitigation measures should therefore concentrate on agricultural and other disturbed systems, but with a focus on the off-site effects. Revegetation on erosion hotspots is therefore the most sustainable way to mitigate soil erosion. Especially grass roots are very effective in controlling concentrated flow erosion (De Baets et al., 2006), which makes indigenous grass species most suitable for revegetation purposes.

7.3. Scale issues in soil erosion research

The multi-scale approach that was used in this thesis enabled a better understanding of soil erosion processes at plot and hillslope scale, which could then be applied in simplified form for soil erosion modelling at catchment scale. In semi-arid environments the non-linear response of runoff and soil erosion can largely be attributed to sinks, which lower the hydrological connectivity. Since these sinks occur at different spatial scales, e.g. vegetation patches at plot scale and agricultural terraces at hillslope scale, the response of runoff and erosion is scale dependent. Upscaling of runoff and erosion measurements should therefore be based on a distributed approach that takes account of these sinks. Using a multi-scale approach requires a combination of different research methods ranging from soil sample analyses and field surveys to remote sensing techniques and models. Combining these different methods, as was adopted in this thesis, will lead to an improved understanding of relevant processes and their effects at broader scales.

Scale has a spatial as well as temporal dimension and together they determine which erosion processes are relevant. The spatial scale in this thesis ranged from plot (1 m^2) to catchment scale (30 km^2) and the temporal scale from minutes to millennia. Figure 7.1 shows at which spatial and temporal scales the relevant erosion and vegetation processes act. Erosion processes that occur at broader scales act in general on longer time scales. However, there are exceptions such as flash floods that occur at catchment scale but within a time frame of a few hours, which make this process so dangerous. The figure also shows that most erosion processes act at a time scale of years or less, while vegetation recovery occurs on a decadal time scale. Land abandonment can therefore lead to an increase of erosion on the short term (1-10 years), but on the longer term the recovery of vegetation and consequent improvement of soil properties will decrease erosion.

An important aspect of temporal scale for soil erosion research is the recurrence time of events, since in semi-arid areas most erosion occurs during only a few large events (Poesen and Hooke, 1997; Mulligan, 1998). Cammeraat (2004) found that the recurrence time of an event that caused large scale terrace failure in a similar catchment as the Carcavo basin is about ten years. Reliable data on recurrence times of extreme events and the effects on soil erosion are needed for realistic risk assessments. Such an assessment in combination with a cost-benefit analysis can be used to determine the sustainability of soil and water conservation practices. The measures proposed to mitigate terrace failure, i.e. revegetation of agricultural terrace walls, might not prevent terrace failure during an extreme event with a recurrence time of more then ten years. Nevertheless, the positive effects of the mitigation measures, e.g. water conservation and soil quality improvement, offset the ineffectiveness of revegetation during extreme events.

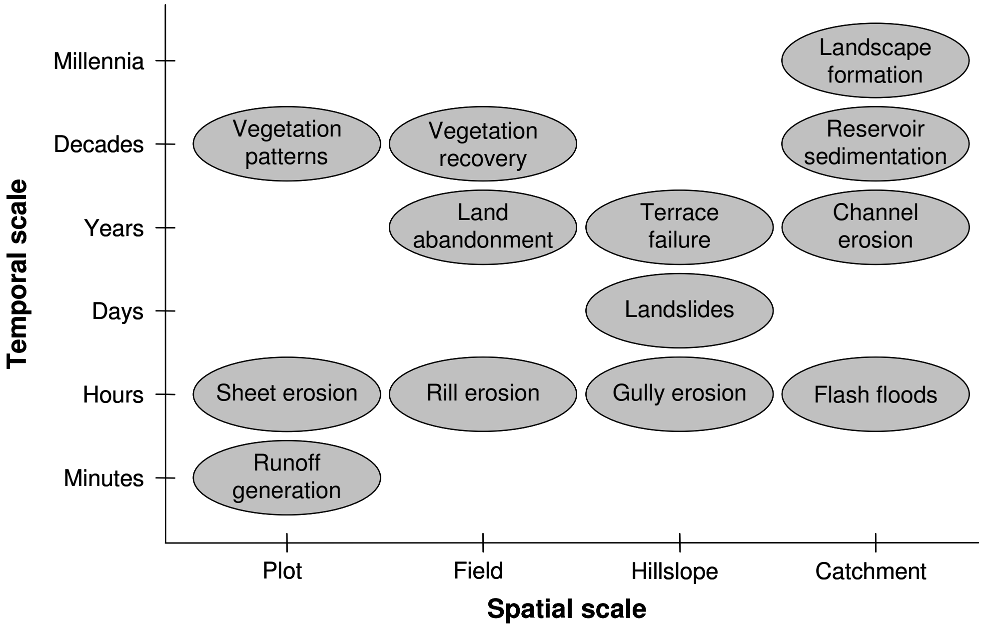


Figure 7.1. Spatial and temporal scales at which relevant erosion and vegetation processes act

Finally, some words should be addressed about the choice of scales for soil erosion research. Often the scale of research and the scale of application do not match, i.e. research is often based on plot or field scale, while the practical application of these results for soil and water conservation programs is often required at catchment scale. To make soil erosion research relevant for policy makers, it should address the scale at which the real decisions are made in terms of soil and water conservation policies (Schulze, 2000). This can be an operational catchment where real decisions on water storage, distribution of irrigation water and water quality are made, and for which hydrological forecasting and soil erosion mitigation are important considerations. Additionally, the objective of a soil erosion study should be clearly defined, since a study to the mitigation of on-site effects asks for another approach and measurement scale than mitigation of off-site effects. Plot scale studies, e.g. based on rainfall simulations, are appropriate to study the on-site effects, while for the off-site effects a distributed modelling approach as applied in this thesis is most suitable.

7.4. Agricultural land abandonment

In marginal areas agricultural land abandonment is frequently occurring and a further increase is expected. Under the semi-arid conditions of the Carcavo basin the consequences in terms of soil erosion are mainly negative. The risk of soil erosion on abandoned fields increases due to slow vegetation recovery, susceptibility to crusting and lack of maintenance of soil and water conservation structures. Especially gully erosion through agricultural terrace walls results in high sediment losses and increases the hydrological connectivity. Besides land abandonment also other land use changes such as agricultural intensification, terraced reforestation and urbanisation will have their drawback on soil erosion processes. These changes in land use often lead to disturbances, which generally enhance soil erosion processes during the first years. Future land use changes should therefore be taken into account when modelling erosion at longer time scales.

As stated before, an increase in agricultural land abandonment in marginal semi-arid areas is expected. For Southeast Spain the main reasons for this increase will probably be water shortage and decreasing EU subsidies. These changes will make the rainfed agriculture economically unprofitable and lead to further land abandonment. Verburg et al. (2006) projected for 2030 that 5 to 13 percent of the total agricultural area in the EU will be abandoned, depending on the different IPCC scenarios. Such large changes will face land use planners and policy makers with the question how to deal with these lands. Leaving abandoned fields completely to nature might lead to land degradation under certain conditions, as demonstrated in this thesis. Nevertheless, with time vegetation is able to recover and vegetation succession will lead to more perennial grasses and shrubs, which will also improve soil quality. However, on highly degraded fields the recovery will be so slow that erosion processes will increase drastically which might lead to irreversible land degradation, i.e. the formation of badlands.

Important soil properties that control soil erosion, such as organic carbon content and aggregate stability, are able to recover to similar levels as under semi-natural vegetation. However, under semi-arid conditions these changes are slow and will take several decades. In more humid areas the vegetation succession may be faster, although disturbances such as grazing will slow down succession as well (Sluiter and De Jong, 2007). Apart from climate also topographic position and soil type determine vegetation succession. Especially the ability of soils to maintain appropriate water availability conditions for successful vegetation establishment is crucial. Recovery rates are higher on north exposed slopes and on slates and slower on south exposed slopes and on marls (Cammeraat et al., 2008), which was also one of the results from Chapter 3. Mitigation measures such as revegetation should therefore be carefully planned taking account of the site specific conditions.

7.5. Recommendations and future research

Hydrological connectivity, distributed modelling of runoff and erosion and a multi-scale approach were main aspects of this thesis. These aspects are important to take into account in soil erosion studies and they deserve further research attention. Runoff and soil erosion modelling should focus more on the spatial patterns, which are very important for the identification of erosion hotspots. Jetten et al. (1999) also stated the importance of calibration by means of comparing observed and simulated runoff and erosion patterns, instead of only total runoff and sediment at the catchment outlet. This means that both modelling and field knowledge should be used for the improvement of runoff and erosion predictions. Connectivity mapping after large events, as applied in the RECONDES project (Hooke, 2003; Hooke and Sandercock, 2007), is a relatively quick methodology to obtain a good impression about the spatial distribution of runoff and erosion. These observed patterns can then be used to calibrate or validate simulated patterns of runoff and erosion. For further improvement of distributed runoff and erosion modelling more detailed DEMs should be used, which better represent topographical features such as small ditches, roads, agricultural terraces and field boundaries, which determine local surface runoff. Nowadays these higher resolution DEMs (1 to 2 m resolution), such as derived from airborne laser altimetry (LiDAR), become increasingly available, and have been successfully applied in other studies, e.g. for gully mapping (James et al., 2007) and flood hydraulics (Cobby et al., 2003). However, the use of higher resolution data will increase the problem of pits and hollows in DEMs, which have to be carefully treated (Lane et al., 2004; Temme et al., 2006).

A multi-scale approach for soil erosion research, as followed in this thesis, allows for the understanding of processes at finer scales and for the assessment of effects at broader (catchment) scales. Such a multi-scale approach requires a combination of research methods of which field investigations should be an important one, since identification of relevant sources and sinks of runoff and erosion can only be done properly in the field. The next step should be upscaling of the obtained field knowledge with GIS and remote sensing techniques, which can be used as input for distributed erosion models. Relatively simple distributed erosion models such as LAPSUS (Schoorl et al., 2002) or Watem/Sedem (Van Oost et al., 2000; Van Rompaey et al., 2001) are most suitable, since more complex models have such a high data demand that use at broader scales becomes nowadays almost impossible. In that case parameters have to be estimated, which will lead to similar or even higher uncertainties as for more simple models. When these models are supplied with more detailed information from high resolution satellite such as QuickBird and IKONOS or LiDAR derived DEMs the prediction accuracy for small catchments will increase substantially.

Finally, some recommendations for potential erosion mitigation strategies within the study area are presented. These strategies are as well representative for semi-arid marginal areas elsewhere in the Mediterranean. In the Carcavo basin the two main hotspots of erosion are gullies through terrace walls and bank gullies along the channel. Both hotspots are often located on the transition of the hillslope to the channel and can have a large impact on the hydrological connectivity. Strategies to mitigate erosion on these hotspots will therefore also reduce the connectivity and prevent sediments to enter the channel. Conservation measures for bank erosion should focus on reducing the amount of runoff draining into active gully heads, which can be achieved by mechanical structures diverting the flow away from the headcut or by adopting land use practices which increase surface roughness and depression storage (Vandekerckhove et al., 2000). In addition, revegetation in channels will reduce the sediment connectivity to downstream areas. For small channels grasses should be planted in areas with fine sediment inputs, while in larger channels, efforts should focus on establishing larger shrubs and trees, which have a greater effect in reducing flow velocities and trapping sediments (Hooke and Sandercock, 2007). Options for mitigation of terrace failure have already been discussed in Chapter 4. Revegetation in zones with concentrated flow near terrace walls is most effective and especially grass species are suitable because of their root properties (De Baets et al., 2006). Potential native species for the Carcavo basin are *Lygeum spartum*, *Brachypodium retusum* and *Stipa tenacissima*. In addition farmers should receive subsidies to maintain abandoned terraces until vegetation has sufficiently established to resist erosion. These subsidies should be combined with workshops on good practices aimed at farmers and land owners and following a bottom-up approach. However, the effectiveness of this kind of revegetation is still unknown, since hardly any studies exist that actually applied revegetation in these semi-arid environments. Besides, other properties such as growth rate, seed germination and the ability to withstand droughts or large events determine whether revegetation will be successful. Further research on these aspects and a cost-benefit analysis of such mitigation measures is needed.

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Summary

The objective of this thesis was to study the interactions between soil, vegetation and erosion in the context of agricultural land abandonment at multiple scales in a semi-arid environment. The research is focused on three central themes, i.e. soil erosion, scale issues and agricultural land abandonment. Soil erosion is one of the main environmental problems in Mediterranean countries which results in soil quality loss and off-site effects such as flash flooding and reservoir sedimentation. For the mitigation of soil erosion it is important to understand the mechanisms and the critical soil conditions that are necessary for maintaining and restoring soil quality. In soil erosion research the issue of scale is very important. Different processes control erosion at the various spatial as well as temporal scales, which leads to different runoff and erosion rates. The spatial scales in this research range from plot to catchment and the temporal scales from minutes to decades. Agricultural land abandonment is one of the main changes in land use in marginal areas of northern Mediterranean countries. However, not much is known about the consequences of land abandonment in terms of land degradation. On the one hand an increase in vegetation cover can decrease erosion, but on the other hand existing soil and water conservation structures are no longer maintained, which can increase erosion. To study these three themes the Carcavo basin was selected as study area. This basin is a catchment in Southeast Spain and is representative for marginal agriculture in semi-arid environments.

In Chapter 2 vulnerable areas for gully erosion were identified using different scenarios of land abandonment. A field survey showed that abandoned fields are more vulnerable to gully erosion compared to cultivated fields. An explanation is the increased runoff concentration on abandoned land due to crust formation and reduced surface storage capacity. The spatial dynamics of land abandonment were simulated with a spatially explicit land use change model for the period 2004 to 2015 for four different land use change scenarios. These results were used to identify vulnerable areas for gully erosion by a simple GIS-model based on the controlling factors of gully erosion. The potentially vulnerable areas for gully erosion increased for all scenarios ranging from 18 ha to 176 ha. The results showed that most of the vulnerable areas are located around channel heads or along channel walls. The combination of a higher gully erosion risk on abandoned fields and an expected increase of land abandonment is potentially a big problem in relation to land degradation and reservoir sedimentation. The identification of vulnerable areas enables soil conservationists and engineers to mitigate gully erosion by applying preventive conservation practices.

In Chapter 3 the development of spatial heterogeneity in vegetation and soil properties after land abandonment was studied. The vegetation composition was described, soil samples collected and detailed aerial photographs were made for two series of abandoned fields on marl and calcrete. These images were classified into bare and vegetated patches, and spatial metrics were calculated for each site. The results showed that recovery of vegetation and changes in soil properties after land abandonment are slow and take at least 40 years under semi-arid conditions. Vegetation succession on calcrete soils appeared to be faster than on marly soils, probably because more water is available on calcrete soils due to a higher rock fragment cover. Organic matter, aggregate stability and electrical conductivity were all significantly higher under vegetated patches. Additionally, a clear linear relationship between vegetation cover and most spatial metrics was found. This relationship was used in Chapter 5 for the upscaling of spotted vegetation patterns.

In Chapter 4 erosion and terrace failure on abandoned land was studied in more detail. At catchment scale all abandoned fields were surveyed and characteristics of each field were described. Additionally abandoned and cultivated terraces were surveyed to determine the factors that induce terrace failure. At field scale a detailed DEM was constructed for an abandoned terrace field to calculate sediment losses since time of abandonment. The results revealed that more than half of the abandoned fields in the Carcavo basin have moderate to severe erosion and the statistical analysis showed that these fields have significantly steeper slopes, are terraced and had cereals as previous land use. Factors that increase the risk of terrace failure were land abandonment, steeper terrace slope, loam texture, valley bottom position and shrubs on the terrace wall. The reconstructed soil erosion rate (87 ton ha^{-1} $year^{-1}$) confirmed the importance of gully erosion on these abandoned terrace fields. Potential soil and water conservation practices to mitigate soil erosion after abandonment are maintenance of terrace walls and revegetation with indigenous grass species on spots with concentrated flow.

Vegetation in semi-arid areas is characterised by heterogeneous patterns of bare and vegetated patches, which has important consequences for biotic and abiotic processes. In Chapter 5 the upscaling of these vegetation patterns, based on QuickBird imagery and detailed aerial photographs, was described. At plot scale detailed aerial photographs were made using a balloon-mounted camera system, which were classified into bare and vegetated patches, to derive the vegetation cover. At catchment scale ten vegetation indices were determined for a high resolution QuickBird image of the study area. These indices were compared with the observed vegetation cover to test which vegetation index had the best correlation. The results showed that most vegetation indices had poor fits with the observed vegetation cover. A simple regression based on the green and red reflectance performed best with a R^2 of 0.91. The low correlations of the other indices are probably related to the atypical spectral behaviour of natural semi-arid vegetation, which is physiologically adapted to dry conditions, and exhibits lower near-infrared reflectances. Consequently, the DVI based indices perform less well, since these are based on the difference between the red and near-infrared reflectance. Determination of vegetation cover in semi-arid environments should therefore include calibration of the selected vegetation index with local field data.

One of the reasons for scale dependency in soil erosion research is the influence of sinks, i.e. areas of infiltration and sedimentation, which lower the hydrological connectivity and decrease the area specific runoff and sediment yield. In Chapter 6 runoff and sediment dynamics were simulated at catchment scale with the LAPSUS model. The model included the effects of plot and hillslope scale features that influence the hydrological connectivity. For the study area vegetation patches and agricultural terraces were the relevant sinks at plot and hillslope scale, respectively. The infiltration module of LAPSUS was elaborated to integrate these runoff sinks by adapting the parameters runoff threshold and runoff coefficient. These parameters were derived from a rainfall simulation database. The results showed that the spatial distribution of vegetation patches and agricultural terraces largely determines the hydrological connectivity at catchment scale. Runoff and sediment yield for the scenario without agricultural terraces were respectively a factor four and nine higher compared to the current situation. Distributed hydrological and erosion models should therefore take account of relevant sinks at finer scales in order to simulate patterns of runoff and erosion correctly at broader scales.

In the last chapter the conclusions from the previous chapters were summarised and the results and implications of this research in relation to the three central themes were discussed. Key aspects of this thesis which deserve further research attention are hydrological connectivity, distributed modelling of runoff and erosion and the multi-scale approach. Connectivity mapping after large events, as applied in the RECONDES project, is a relatively quick methodology to obtain a good overview of the spatial distribution of runoff and erosion. These patterns can then be used to calibrate or validate distributed erosion models. Runoff and soil erosion modelling should focus more on spatial patterns, which are important for the identification of erosion hotspots. A multi-scale approach for soil erosion research allows for the understanding of processes at finer scales and for the assessment of effects at broader (catchment) scales.

Samenvatting

Het doel van dit proefschrift was het bestuderen van de interacties tussen de bodem, vegetatie en erosie op verschillende schalen in de context van landverlating in een semi-aride gebied. Het onderzoek richtte zich op drie centrale thema's, namelijk bodemerosie, schaal en het verlaten van landbouwgronden. Bodemerosie is één van de belangrijkste milieuproblemen in Mediterrane landen en leidt tot het verlies van bodemkwaliteit en kan ook leiden tot overstromingen en sedimentatie in stuwmeren. Om bodemerosie tegen te gaan is het belangrijk om de mechanismen en de kritische bodemcondities die nodig zijn voor het behouden en herstellen van de bodemkwaliteit te begrijpen. In bodemerosie-onderzoek is schaal een belangrijk begrip. Op de diverse ruimtelijke en temporele schalen zijn er verschillende processen die de erosie beïnvloeden, wat leidt tot verschillende erosiesnelheden. De ruimtelijke schalen binnen dit onderzoek variëren van plot tot stroomgebied en de temporele schalen van minuten tot decennia. Verlating van landbouwgronden is één van de voornaamste veranderingen in landgebruik in marginale regio's van Europese mediterrane landen. Er is echter niet veel bekend over de gevolgen van landverlating in relatie tot landdegradatie. Aan de ene kant leidt een toename in vegetatie tot vermindering van de erosie, maar aan de andere kant worden systemen voor bodemconservering niet langer onderhouden, wat kan leiden tot een toename van erosie. Om deze drie thema's te bestuderen is het Carcavo stroomgebied gekozen als studiegebied. Dit stroomgebied is gelegen in zuidoost Spanje en is representatief voor marginale landbouw in semi-aride gebieden.

In hoofdstuk 2 werden kwetsbare gebieden voor geulerosie geïdentificeerd voor verschillende scenario's van landverlating. Uit veldonderzoek bleek dat verlaten velden kwetsbaarder zijn voor geulerosie dan gecultiveerde velden. Dit kan verklaard worden door de toename van 'runoff' vanwege korstvorming en een afname van de waterbergingscapaciteit van de bodem. De ruimtelijke dynamiek van landverlating werd gesimuleerd met een ruimtelijk expliciet landgebruiksveranderingsmodel voor de periode 2004 tot 2015 en voor vier verschillende scenario's. De resultaten werden gebruikt om kwetsbare gebieden voor geulerosie te identificeren door gebruik te maken van een simpel GIS-model gebaseerd op de factoren die geulerosie beïnvloeden. Het areaal potentieel kwetsbare gebieden voor geulerosie nam in alle scenario's toe, variërend van 18 tot 176 hectare. De resultaten lieten zien dat de kwetsbare gebieden voornamelijk zijn gelegen langs de ingesneden droge rivierbeddingen. De combinatie van een groter risico op geulerosie op verlaten velden en de verwachte toename van landverlating is potentieel een groot probleem vanwege toegenomen landdegradatie en sedimentatie in de stuwmeren. De

identificatie van de kwetsbare gebieden voor geulerosie maakt het voor beheerders mogelijk om preventieve bodemconserveringsmaatregelen te treffen.

In hoofdstuk 3 werd de ontwikkeling van ruimtelijke heterogeniteit van vegetatie en bodemeigenschappen na landverlating bestudeerd. De samenstelling van de vegetatie werd beschreven, bodemmonsters werden verzamelden en gedetailleerde luchtfoto's werden gemaakt voor twee series van verlaten velden, één op mergel en één op 'calcrete'. Deze luchtfoto's werden geclassificeerd in kale grond en vegetatie en voor elke plot werden de ruimtelijke metrieken berekend. De resultaten lieten zien dat herstel van de vegetatie en veranderingen in bodemeigenschappen na landverlating langzaam gaan en onder de semi-aride omstandigheden van het studiegebied minstens 40 jaar duren. De vegetatiesuccessie op calcrete bodems lijkt sneller te gaan dan op mergelbodems, waarschijnlijk omdat in de calcrete bodems meer water beschikbaar is door de vele stenen aan het oppervlak. Het organisch stof gehalte, de aggregaatstabiliteit en de elektrische geleidbaarheid waren allemaal significant hoger onder de vegetatie. Daarnaast werd een duidelijke lineaire relatie tussen de vegetatiebedekking en de meeste ruimtelijke metrieken gevonden. Deze relatie werd later in hoofdstuk 5 gebruikt voor het opschalen van vegetatiepatronen.

In hoofdstuk 4 werden erosie en terrasdoorbraken op verlaten velden in meer detail bestudeerd. Alle verlaten velden binnen het studiegebied werden in kaart gebracht en de eigenschappen van elk veld werden beschreven. Ook werden terrassen van verlaten en gecultiveerde velden gekarteerd en beschreven om te onderzoeken welke factoren de doorbraak van terrassen bepalen. Op veldniveau werd een gedetailleerd digitaal hoogte model gemaakt om de erosie sinds het moment van verlating te bepalen. De resultaten lieten zien dat meer dan de helft van de verlaten velden in het studiegebied matige tot ernstige erosie hebben. Uit de statistische analyse volgde dat deze velden significant steilere hellingen hebben, geterrasseerd zijn en granen als vorig landgebruik hadden. Factoren die het risico van terrasdoorbraken bevorderen waren: landverlating, een steilere helling van het terras, een leemtextuur, een dalpositie en struiken op de terraswal. De gereconstrueerde bodemerosie snelheid (87 ton ha^{-1} $jaar^{-1}$) bevestigde het belang van geulerosie op deze verlaten geterrasseerde velden. Potentiële bodem- en waterconserveringsmaatregelen om bodemerosie tegen te gaan zijn het onderhouden van terraswallen en hervegetatie met inheemse grassoorten op plaatsen waar runoff zich concentreert.

Vegetatie in semi-aride gebieden wordt gekarakteriseerd door heterogene patronen van kale grond en vegetatie die belangrijke gevolgen hebben voor biotische en abiotische processen. In hoofdstuk 5 werd het opschalen van deze vegetatiepatronen met behulp van QuickBird satellietbeelden en gedetailleerde luchtfoto's beschreven. Op plotschaal werden gedetailleerde luchtfoto's gemaakt met behulp van een camerasysteem gemonteerd aan een ballon gevuld met helium. Deze luchtfoto's werden geclassificeerd in kale grond en vegetatie om de vegetatiebedekking af te leiden. Op stroomgebiedschaal werden tien vegetatie-indexen afgeleid van een QuickBird beeld van het studiegebied. Deze indexen werden vergeleken met de geobserveerde vegetatiebedekking om te testen welke vegetatie-index de hoogste correlatie had. De meeste vegetatie-indexen hadden een slechte 'fit' met de geobserveerde vegetatiebedekking. Een eenvoudige regressie gebaseerd op de groene en rode reflectie had de beste fit met een R^2 van 0,91. De slechte correlaties van de andere indexen zijn waarschijnlijk gerelateerd aan het afwijkende spectraal gedrag van semi-aride natuurlijke vegetatie. Deze is fysiologisch aangepast aan de semi-aride omstandigheden en heeft daardoor lagere nabij-infrarood reflecties. Hierdoor zijn de DVI gebaseerde indexen minder geschikt, omdat deze gebaseerd zijn op het verschil tussen de rode en nabij-infrarode reflectie. Voor de bepaling van de vegetatiebedekking in semi-aride gebieden zou de gekozen vegetatie-index altijd gekalibreerd moeten worden met lokale velddata.

Eén van de redenen voor schaalafhankelijkheid in bodemerosieonderzoek is de invloed van 'sinks', plekken waar extra water infiltreert en/of sedimentatie plaats vindt. Deze sinks verlagen de hydrologische connectiviteit en verminderen de gebiedsspecifieke runoff en sedimentopbrengst. In hoofdstuk 6 werden runoff en sedimentdynamiek op stroomgebiedschaal gesimuleerd met het LAPSUS model. De invloed van plot- en hellingschaal eigenschappen die de hydrologische connectiviteit beïnvloeden zijn daarbij meegenomen. Voor het Carcavo studiegebied waren vegetatiepatronen en landbouw-terrassen de relevante sinks op respectievelijk plot en hellingschaal. De infiltratiemodule van LAPSUS werd uitgebreid met een 'runoff drempel' en een 'runoff coëfficiënt' om de invloed van deze runoff sinks te integreren in het model. Deze coëfficiënten werden afgeleid uit een database van regensimulaties. De resultaten laten zien dat de ruimtelijke verdeling van vegetatiepatronen en landbouwterrassen voor een belangrijk deel de hydrologische connectiviteit bepalen op stroomgebiedschaal. Runoff en sediment opbrengst voor het scenario zonder landbouwterrassen respectievelijk vier en negen keer hoger vergeleken met de huidige situatie. Ruimtelijke hydrologische en erosiemodellen zouden daarom rekening moeten houden met de relevante sinks op de onderliggende schaalniveaus, zodat runoff en erosie patronen correct gesimuleerd worden op grotere schaal.

In het laatste hoofdstuk van dit proefschrift werden de conclusies van de vorige hoofdstukken samengevat en de resultaten en implicaties van het onderzoek in relatie tot de drie centrale thema's bediscussieerd. Belangrijke aspecten van dit proefschrift die nader onderzoek verdienen zijn: hydrologische connectiviteit, ruimtelijke modellering van runoff en erosie en de multi-schaal benadering. Het in kaart brengen van de hydrologische connectiviteit na grote buien, zoals toegepast in het RECONDES project, is een relatief snelle methode om een goed overzicht te verkrijgen van de ruimtelijke verdeling van runoff en erosie. Deze patronen kunnen dan gebruikt worden voor het kalibreren of valideren van ruimtelijke erosiemodellen. Het modelleren van runoff en erosie zou zich meer moeten focussen op de ruimtelijke patronen die belangrijk zijn voor de identificatie van erosie hotspots. Een multi-schaal benadering voor erosieonderzoek zorgt er voor dat de processen op fijnere schaal beter begrepen worden en dat de effecten op grotere (stroomgebied) schaal beter ingeschat worden.

Resumen

El objetivo de esta tesis fue estudiar la interacción entre el suelo, la vegetación y la erosión a escalas múltiples en el contexto de abandonamiento de tierras agrícolas en una región semiárida. La investigación se enfocó en tres temas centrales: la erosión del suelo, cuestiones de escalas y el abandonamiento de las tierras agrícolas. La erosión del suelo es uno de los principales problemas ambientales en los países mediterráneos y se manifiesta con la pérdida de la calidad del suelo y produce tales efectos como inundaciones repentinas y sedimentación en embalses. Para la mitigación de la erosión es importante comprender los mecanismos y las condiciones críticas del suelo, que son necesarias para el mantenimiento y la restauración de la calidad del suelo. En la investigación de la erosión, la noción de escala es muy importante. En varias escalas espaciales y temporales están involucrados diferentes procesos que afectan la erosión, lo que da lugar a diferentes tasas de escorrentía y de erosión. Las escalas espaciales en esta investigación van desde parcela hasta cuenca y las escalas temporales de minutos a décadas. El abandono de las tierras agrícolas es uno de los principales cambios en el uso del suelo en las zonas marginales de los países del Mediterráneo septentrional. Sin embargo, no se sabe mucho sobre las consecuencias del abandonamiento de las tierras en términos de degradación del suelo. Por un lado un aumento de vegetación puede reducir la erosión, pero por otro lado ya no son mantenidos los sistemas de conservación del suelo, que puede incrementar la erosión. Para estudiar estos tres temas, fue seleccionada como área de estudio la cuenca del Carcavo. Esta cuenca se encuentra situada al sureste de España y es representativa de la agricultura marginal en zonas semiáridas.

En el capítulo 2, se identificaron zonas vulnerables a la erosión en cárcavas para los diferentes escenarios de abandonamiento de las tierras. Un estudio de campo reveló que los campos abandonados son más vulnerables a la erosión en cárcavas, frente a los campos de cultivo. Una explicación es el aumento de la concentración de la escorrentía en las tierras abandonadas debido a la formación de costras y a la reducción de capacidad de retención de agua en el suelo. La dinámica espacial del abandonamiento de las tierras fue simulada con un modelo explícito espacial de cambio del uso de la tierra. La simulación fue en el período entre 2004 y 2015 y por cuatro diferentes escenarios. Estos resultados se utilizaron para identificar las zonas vulnerables a la erosión de cárcavas con un simple SIG-modelo basado en los factores que influencian la erosión de cárcavas. Las superficies potencialmente vulnerables para el aumento de la erosión de cárcavas aumentaron en todos los escenarios, que van desde 18 hectáreas a 176 hectáreas. Los resultados mostraron que las zonas vulnerables se encuentran principalmente a lo largo de las regiones de las ramblas. La

combinación de un mayor riesgo de erosión de cárcavas en campos abandonados y un aumento previsto del abandonamiento de las tierras es potencialmente un gran problema a causa de la degradación del suelo y sedimentación en embalses. La identificación de las zonas vulnerables, permite que los conservacionistas y los ingenieros mitiguen la erosión de cárcavas mediante la aplicación de prácticas de conservación preventiva.

En el capítulo 3, se describe el estudio del desarrollo de heterogeneidad espacial en la vegetación y las propiedades del suelo tras el abandono de las tierras. Se describieron tambien la composición de la vegetación, muestras de suelo fueron recogidas y se hicieron detalladas fotografías aéreas de dos series de campos abandonados, una en margas y la otra en 'calcrete'. Estas fotos aéreas se clasificaron en suelo desnudo y vegetación y se calcularon las métricas espaciales para cada sitio. Los resultados mostraron que la recuperación de la vegetación y los cambios en las propiedades del suelo tras el abandono de las tierras son lentos y en condiciones semiáridas duran al menos 40 años. Sucesión de vegetación en suelos calcrete parecía ser más rápido que en los suelos margosas, probablemente porque en los suelos calcrete mas agua está disponible debido a una mayor cobertura de fragmento de rocas. La materia orgánica, la estabilidad de agregados y la conductividad eléctrica fueron significativamente mayores debajo de la vegetación. Además, se encontró una clara relación lineal entre la cubierta vegetal y la mayoría de métricas espaciales. Esta relación fue utilizada en el capítulo 5 para la ampliación de los patrones de vegetación.

En el capítulo 4 se estudió más detalladamente la erosión y la rotura de terrazas en tierras abandonadas. Todos los campos abandonados dentro del área de estudio fueron encuestados y las características de cada campo fueron descritas. Además fueron encuestadas las terrazas abandonados y cultivadas para determinar los factores que inducen a la rotura de terrazas. También se construyó un modelo digital de elevación detallado, para determinar la erosión desde el momento del abandono. Los resultados revelaron que más de la mitad de los campos abandonados en la cuenca de Carcavo sufren de una moderada a severa erosión. El análisis estadístico demostró que estos campos tienen pendientes significativamente más inclinados, con terrazas y con cereales como anterior uso de la tierra. Factores que aumentan el riesgo de rotura de terrazas fueron: el abandono de las tierras, una pendiente más inclinada de la terraza, una textura limosa, una posición de valle y arbustos en la pared de la terraza. La tasa de erosión reconstruida (87 toneladas ha^{-1} $año^{-1}$) confirmó la importancia de la erosión de cárcavas en estos campos abandonados con terrazas. Potenciales prácticas de conservación de suelo y agua para mitigar la erosión son el mantenimiento de paredes de terrazas y revegetación con especies autóctonas de gramínea en lugares con flujo concentrado.

Vegetación de zonas semiáridas se caracteriza por patronos heterogéneos de suelo desnudo y vegetación, que tiene consecuencias importantes para procesos bióticos y abióticos. En el capítulo 5, se describió la ampliación de estos patrones de vegetación, basado en una imagen QuickBird y detalladas fotografías aéreas. A nivel de parcela, se realizaron detalladas fotografías aéreas mediante un sistema de cámara montado en un globo. Estas fueron clasificadas en suelo desnudo y vegetación, para obtener la cobertura vegetal. A nivel de cuenca se determinaron diez índices de vegetación de una alta resolución imagen QuickBird de la zona de estudio. Estos índices fueron comparados con la cubierta vegetal observada, para probar que índice de vegetación tuvo el mayor coeficiente de correlación. Los resultados mostraron que la mayoría de índices de vegetación tenían una correlación baja con la cubierta vegetal observada. Una simple regresión basada en la reflexión verde y roja tenía la mejor correlación con un R^2 de 0,91. Las bajas correlaciones de los otros índices son probablemente relacionados con el comportamiento espectral atípico de una vegetación natural semiárida. Esta vegetación esta fisiológicamente adaptada a las condiciones secas y muestra una reflexión inferior del infrarrojo cercano. En consecuencia, los índices basados en DVI son menos apropiados, ya que estos se basan en la diferencia entre la reflexión del rojo y del infrarrojo cercano. Determinación de la cubierta vegetal en zonas semiáridas debe incluir la calibración de los índices de vegetación seleccionados con datos de campo local.

Una de las razones para la escala de dependencia en investigación de erosión es la influencia de los ‘sinks’, es decir, zonas de infiltración y de sedimentación, que reducen la conectividad hidrológica y disminuyen el área específica de escorrentía y rendimiento del sedimento. En el capítulo 6 se simuló la dinámica de escorrentía y sedimento a nivel de las cuencas con el modelo LAPSUS. El modelo incluyó los efectos de las características de la escala de la parcela y de la pendiente que influyen a la conectividad hidrológica. Para el área de estudio los ‘sinks’ relevantes fueron patrones de vegetación y terrazas agrícolas. El módulo de infiltración de LAPSUS fue elaborado para integrar esos ‘sinks’ de escorrentía mediante la adaptación de los parámetros umbral de escorrentía y coeficiente de escorrentía. Estos parámetros se obtuvieron de una base de datos a partir de simulaciones de lluvia. Los resultados mostraron que la distribución espacial de los patrones de vegetación y terrazas agrícolas determina en gran medida la conectividad hidrológica a nivel de la cuenca. La escorrentía y rendimiento del sedimento para el escenario sin terrazas agrícolas fueron respectivamente un factor cuatro y nueve más alto en comparación a la situación actual. Modelos de erosión distribuidos deben tomar en cuenta los ‘sinks’ de las escalas más pequeñas, con el fin de simular patrones de escorrentía y erosión correctamente a escalas más grandes.

En el último capítulo se resumieron las conclusiones de los capítulos anteriores y se debatieron los resultados y las implicaciones de esta investigación en relación con los tres temas centrales. Los aspectos claves de esta tesis que merecen más atención son: conectividad hidrológica, modelos de escorrentía y erosión distribuidos y la visión multi-escala. Cartografía de la conectividad hidrológica después de grandes eventos, tal como se aplicaron en el proyecto RECONDES, es un método relativamente rápido para obtener una visión general buena de la distribución espacial de la escorrentía y la erosión. Estos patrones se pueden utilizar para calibrar o validar modelos de erosión distribuidos. Los modelos de escorrentía y erosión deberían centrarse más en los patrones espaciales, que son importantes para identificar los puntos críticos de la erosión. Una visión multi-escala para la investigación de la erosión permite la comprensión de los procesos a escalas más pequeñas y para la evaluación de los efectos en escalas más grandes.

Acknowledgements

The writing of a PhD thesis is not the work of only one person, but the result of cooperation with many other people. Therefore I would like to thank everyone who in one way or another contributed to this thesis. First of all I would like to thank Koos Verstraten for being my promoter. Although we did not have very intensive contact during most of the PhD, you were always interested in my work and arranged the necessary conditions for doing good research. Many thanks I own to my co-promoter Erik Cammeraat. Erik, you introduced me very well into the world of soil erosion and learned me a lot about erosion monitoring and laboratory measurements. You also gave me a lot of freedom to come with my own ideas and make my own plans, which I highly appreciated.

This PhD was part of the RECONDES project, a three year international project financed by the European Commission. I very much enjoyed the collaboration with the people of the RECONDES team. I would like to thank Janet Hooke of the University of Portsmouth for coordinating the project perfectly and in such a way that collaboration was stimulated and people were made enthusiastic. Peter Sandercock thanks for all your GPS surveys and Miguel Marchamalo for the connectivity surveys. Our Spanish partners of CSIC-CEBAS introduced us to the Carcavo basin, assisted us with plant species determination, downloaded our monitoring data and arranged nice restaurants for the RECONDES meetings. Victor Castillo, Gonzalo Barberá, José Antonio Navarro, Carolina Boix Fayos and José Ignacio Querejeta thank you all for that. Our colleagues Bas van Wesemael and André Meerkerk of the Université Catholique de Louvain are thanked for their collaboration regarding the balloon photography, connectivity mapping and rainfall simulations. I also enjoyed working with Jean Poesen and Sarah de Baets of the Katholieke Universiteit Leuven and Dino Torri, Lorenzo Borselli, Michela Marignani, Pilar Salvador of CNR IRPI in Italy.

The most important part of a PhD is writing papers, therefore I would like to acknowledge the co-authors of my papers. Erik Cammeraat, Kasper Kok, Peter Verburg, Annemieke Kooijman, Bas van Wesemael, Harry Seijmonsbergen, Janet Hooke and Jeroen Schoorl thank you all for your constructive comments and suggestions, which greatly improved the quality of the papers. Also the anonymous reviewers of the papers are acknowledged for their useful suggestions and corrections.

A special thank I owe to Francisco Garcia and Gregorio, who allowed me to install monitoring equipment on their land and to store our equipment in their farm. Besides, I was always warmly invited to join their typical Spanish lunches on Sundays. I also thank Inmaculate Castellón of the Cooperation in Planes and Tomas Navarro Martinez from Baños de Fuen Santo for the storage of our equipment.

Wouter van Gorp, Elmer Cladder, Susanne Vijfhuizen, Margriet Achtereekte, Tim Nieman, Tom Loring and Richard Kanbier were all students who did their master thesis research or geo-ecology course within my PhD research. Thank you all for the collection of very useful data and I enjoyed doing fieldwork with you. I also want to express thanks to the people of the laboratory, Leo, Ton, Piet, Bert, Leen and Joke, for analysing my soil samples and for your assistance when I was doing analyses on my own.

Although we did not move to Amsterdam when I started this PhD, my colleagues at the UvA familiarised me with the university and the city. First of all I want to thank Femke for being one of my paranimfs and for all our talks about the struggles of a PhD and all kinds of other subjects. Also thanks to Boris, Klaas, Sanneke and Maartje with whom I spent my lunch breaks and had nice conversations, and to my roommates Klaasjan, Jasper, Ulrich, Klaske and Alejandro.

Also my friends and family indirectly contributed to this thesis by showing their interest and providing the necessary social distraction. I want to thank Ronald for being one of my paranimfs and for spending part of his holiday digging out cables and sensors, Roel and Noud for climbing the Almorchon with me, and Thomas for giving me some useful tips about the printing of a thesis. I want to thank my parents for all their support and the interest they always showed for my research. They even came to visit and help me during one of my fieldworks in Spain.

The last words of gratitude are reserved for Irina and Rubén. Querida Irina, quiero agradecerte mucho por tu paciencia y apoyo durante estos cuatro años. Yo se que no te gustó mucho mis viajes frecuentes al extranjero, pero si me diste la ayuda para terminar este doctorado con éxito. Y finalmente para mi querido Rubén unos palabras, tu nacimiento me dió mucha felicidad y energía para terminar rápido esta tesis. Aunque no haya podido darte tanta atención en este periodo como quería, desde ahora voy a pasar más tiempo contigo.

Curriculum Vitae

Jan Peter Lesschen was born on 8 June 1978 in Oosterhesselen, a small village in the province of Drenthe. There he grew up and went to primary school. He finished his secondary school (VWO) at the Nieuwe Veste in Coevorden. Then he decided to study Soil, Water en Atmosphere at Wageningen University. For his specialisation soil science he conducted two thesis researches. In his first thesis he reconstructed and modelled the course of the Guadalhorce river during the Pliocene, for which he did fieldwork in South Spain. For his second thesis he went to Ecuador where he investigated the influence of soil variability for the Tradeoff Analysis Model in the potato area of northern Ecuador. In March 2002 he graduated 'cum laude' for his Masters degree.

Jan Peter started working as junior researcher at the Laboratory of Soil Science and Geology of Wageningen University for a FAO-commissioned study "Scaling soil nutrient balances". After this project he remained working at the Laboratory of Soil Science and Geology as assistant lecturer to develop new courses, and as research assistant to develop documentation and exercises for the CLUE-S model and to document statistical methods for analysing spatial patterns. After these research jobs he decided to go for a PhD degree. In April 2004 he started a PhD at the Institute for Biodiversity and Ecosystem Dynamics of the University of Amsterdam.

His PhD project was part of the RECONDES project (Conditions for restoration and mitigation of desertified areas using vegetation), a three year international research project funded by the EU. During his PhD he supervised several Master students and contributed to a number of Bachelor courses. He also followed the courses of the research school ICG (Centre for Geo-ecological research). In 2008 he finished his thesis titled "Multi-scale interactions between soil, vegetation and erosion in the context of agricultural land abandonment in a semi-arid environment". Since April 2008 Jan Peter is working as researcher at Alterra in Wageningen. There he is involved in projects related to greenhouse gas emissions and integrated nitrogen modelling and he is responsible for the MITERRA-Europe model. Jan Peter is married to Irina and they have a son named Rubén.