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0.2 Notice

Parts of this thesis have been published elsewhere or have been submitted to scientific journals, as indicated below.

Chapter 2:

European Commission, 2007. Conditions for restoration & mitigation of desertified areas using vegetation (RECONDES) - Review of literature and present knowledge. European Commission, Brussels. ISBN: 92-79-03072-8.

Chapter 3:

Meerkerk, A.L., van Wesemael, B., Cammeraat, E., *accepted*. Water availability in almond orchards on marl soils in south east Spain: The role of evaporation and runoff. Journal of Arid Environments.

Chapter 5:

Meerkerk, A.L., van Wesemael, B., Bellin, N., *submitted*. Overland flow in semi-arid catchments: the impact of terrace failure on hydrological connectivity. Hydrological Processes.

Chapter 4 & 6 are expected to be submitted later on.

0.3 Acknowledgements

Many people have contributed directly or indirectly to the achievement of this Ph.D. thesis. I am very grateful for their support over the past four years, which includes the exchange of ideas and information, discussions, advice and feedback, assistance and collaboration during field campaigns, laboratory work, data analysis, etc. In particular I would like to thank the following people:

Bas van Wesemael, my supervisor, for putting a great deal of energy into the research and papers. His sense for research opportunities, excellent knowledge of the English language, proficiency to put things into perspective, as well as his experience in terms of field work and lysimeter studies have been important inputs to the thesis and to the development of my professional skills. Furthermore I highly appreciate that he is available for his students at all times.

Erik Cammeraat and Jan-Peter Lesschen for their collaboration, especially in the field, providing us with a rainfall simulator, theodolite, meteorological data and soil sample analysis.

Gonzalo González Barberá and his colleagues of the Centro de Edafología y Biología Aplicada del Segura (CEBAS-CSIC), Murcia, Spain, for providing valuable assistance in terms of field work logistics and data on the almond production in Murcia.

José Alfonso Gómez (CSIC Córdoba), Juan Vicente Giráldez (University of Córdoba) and their colleagues, who enabled us to make soil moisture measurements in an olive orchard near Sevilla, Spain, and who shared their weather and soil data (Chapter 6).

The members of the jury (§0.1) for their availability and comments, which form a significant contribution to the quality of the thesis.

Peter Sandercock for double checking the replacement of my first flat car tire during field work in Cárcavo. Furthermore for his great contribution and commitment to the RECONDES (Conditions for restoration and mitigation of desertified areas using vegetation) project.

Marco Bravin for his assistance with the lysimeter experiments and field work preparations.

Nicolas Bellin for the joined field work and the cooperation in the study of hydrological connectivity.

The other members of the RECONDES team, in particular Janet Hooke and Sarah De Baets.

Other members of the "Connectivity team": Olivier Evrard, Elisabeth Frot, Michaël Antoine.

Other colleagues at the Department of Geography of the UCL.

A number of colleagues that provided us with information on the cultivation of rainfed tree crops throughout the Mediterranean (Chapter 4): A. Almeida, F. Dicenta, M. Marignani, A. Pinheiro, A. Tubeileh.

Isabel Baeza Varela for her inspiring Spanish classes.

I would like to express my gratitude to the European Commission (Directorate General of Research, Global Change and Desertification Programme, Project No. GOCE-CT-2003-505361) and the Fonds Spéciaux de Recherche de la Communauté Française de Belgique for funding the thesis.

I would like to thank Syngenta Ltd. for their support of the ProTerra project and the field experiments at Benacazón.

0.4 Contents

0.5 List of symbols and abbreviations

A	catchment area [km^2]
A_c	contributing area of concentrated flow to the river [ha]
A_Q	area producing concentrated flow [ha]
ANOVA	analysis of variance
ATEAM	Advanced Terrestrial Ecosystem Analysis and Modelling (EU Project No. EVK2-2000-00075)
$\alpha(h)$	contributing area function [-]
C	coefficient of runoff [mm mm^{-1}]
CAP	Common Agricultural Policy
CC	cover crop treatment
CN	curve number
CREM	Centro Regional de Estadística de Murcia
DEM	digital elevation model
D	drainage of soil water [mm]
D_{15}	drainage of water from the upper 15 cm of the soil [mm]
DOY	day of year
ΔS	change in soil moisture storage over a given time interval [mm]
E_0	potential evaporation [mm]
E_k	rainfall kinetic energy [J m^{-2} mm^{-1}]
E_p	evaporation from the plough layer [mm]
E_s	evaporation from the entire soil [mm]
EC	European Commission

EI_{30}	storm erosivity index [MJ ha^{-1} mm h^{-1}]
ET_0	reference crop evapotranspiration [mm]
ET_a	actual crop evapotranspiration [mm]
ET_c	potential crop evapotranspiration [mm]
EU	European Union
FAO	Food and Agricultural Organisation
GPS	global positioning system
h	lag distance [m]
HI	humidity index
HI_{dry}	average humidity index for years that fall within the 33rd percentile of annual rainfall
HRU	hydrological response unit
IPCC	Intergovernmental Panel on Climate Change
I	infiltration of water into the soil [mm]
I_{10}	ten minute peak intensity of a rainstorm [mm h^{-1}]
I_a	initial abstraction [mm]
I_α	integral connectivity scale of the $\alpha(h)$ function [m]
I_ψ	integral connectivity scale of the $\psi(h)$ function [m]
I_τ	integral connectivity scale of the $\tau(h)$ function [m]
K_c	crop coefficient [-]
K_r	soil evaporation reduction coefficient [-]
K_{sat}	saturated hydraulic conductivity [mm h^{-1}]
L	length of the longest flow path to the catchment outlet [m]

MAPA	Ministerio de Agricultura, Pesca y Alimentación
n	number of observations
p	probability of a type I error in the context of a statistical test [-]
pF	${}^{10}\log(-h_m/\text{cm})$ [-], where h_m [cm] is the matric head of soil water (Koorevaar et al., 1983)
P	precipitation [mm]
P_{50}	precipitation during the 50 minutes outside the 10 minute rainfall peak of an hour of rainfall [mm]
$\psi(h)$	hydrological connectivity function [-]
Q_{off}	runoff water [mm]
Q_{on}	run-on water [mm]
Q_p	peak discharge [m^3 s^{-1}]
r^2	coefficient of determination [-]
R	rainfall intensity [mm h^{-1}]
R_a	extraterrestrial radiation [mm a^{-1}]
S	slope gradient [-]
SRES	Special report on emission scenarios
SWC	soil and water conservation
t_{50}	the time of precipitation during the 50 minutes outside the 10 minute rainfall peak of an hour of rainfall [h]
T	return period [year]
T_c	time to concentration [min]
TI	tillage treatment
T_{max}	maximum air temperature [ºC]
T_{mean}	mean air temperature [ºC]

T_{min}	minimum air temperature [ºC]
θ_i	initial soil moisture [m^3 m^{-3}]
τ_d	drainage characteristic [-]
$\tau(h)$	connectivity function [-]
UNEP	United Nations Environment Programme
USA	United States of America
WUE	water use efficiency [kg mm^{-1} ha^{-1}]

CHAPTER 1
INTRODUCTION

1.1 Background

The world population has increased six fold between 1800 and 2000, and is expected to grow another 40% by 2050, reaching 8.9 billion (UN-ESA, 2000). At the same time, the scope for increasing the area of agricultural land and the use of renewable fresh water resources is limited, so that food and water shortages are expected to increase (Wallace, 2000). It has been predicted that by 2050, one in six of the world population will have insufficient water to meet their basic requirements and 67% of the population will experience some level of water shortage (Wallace, 2000). Furthermore, global warming leads to increased aridity and threatens the long-term agricultural productivity in semi-arid areas like the Mediterranean region, the western part of the grain belt of the United States and California. The prevention of dryland degradation has therefore become a high priority (UNEP, 1997; MEA, 2005). Another trend that promotes food scarcity and prices, is the increase in the production of biofuels (Kruse et al., 2007; Rajagopal, 2008).

One effect of this pressure on water and soil resources is the need to expand the agricultural area. It is therefore likely that more marginal lands, like steep slopes, soils of low productivity and drylands will be taken into cultivation (e.g. Doolittle, 1983; Soini, 2005). Combined with the intensification of agricultural land use, this means that the area prone to land degradation increases (De Graaff and Eppink, 1999; Faulkner et al., 2003; van Wesemael et al., 2006). One strategy to mitigate dryland degradation and maintain yield levels is to prevent accelerated soil erosion using soil conservation measures (MEA, 2005). Another strategy is to increase the water use efficiency of the cropping system, which can be defined as the amount of biomass or harvestable product produced per unit of water (Lövenstein et al., 1995; Wallace, 2000).

1.2 Research topic

This thesis focuses on rainfed orchards in the Mediterranean region. These cropping systems are characterised by a low soil cover during a large part of the year and intensive tillage (Poesen et al., 1997; Borselli et al., 2002; Pastor, 2004). As a result, the tree plantations are associated with high rates of soil erosion, especially in hilly and mountainous areas (De Graaff & Eppink, 1999; Borselli et al., 2002; Beaufoy, 2003; Gómez et al., 2005). Studies of global warming foresee higher air temperatures and a greater interannual climatic variability (Olesen & Bindi, 2002). Higher air temperatures reduce the length of the growing season, which may shorten the period during which the soil is adequately protected against erosion. A larger interannual rainfall variability will increase yield fluctuations and the risk of crop failure. Hence there is a need for policies and measures to address and mitigate the degradation of agricultural lands.

A wide range of soil and water conservation (SWC) measures have been documented and applied in semi-arid areas over the past decades (e.g. Hudson, 1985; Ennabli, 1993; Morgan, 1995; van Wesemael et al., 1998; Rockström, 2003; Yuan, 2003; Castillo et al., 2007; Frot et al., 2008). Traditional SWC measures collect runoff water behind terraces and dams. Despite their beneficial effects, there has been an overall abandonment of terraces in the Mediterranean (Grove & Rackham, 2003). This is most probably explained by the costs associated with the construction and maintenance of SWC measures. It appears that from a producer's perspective, it is more economical to remove terraces and install irrigated orchards on low-priced drylands, than to maintain some form of conservation measures (Oñate & Peco, 2005). Furthermore, there has been a shift in cultivation practices over the last decades, aiming to increase the size of agricultural fields, intensify the production and facilitate mechanisation (Borselli et al., 2002; Beaufoy, 2003). The trend of mechanisation and the expansion of tree crops has been influenced by agricultural policy (De Graaff & Eppink, 1999; Olesen & Bindi, 2002). It has been demonstrated that the Common Agricultural Policy of the EU has promoted the expansion of olive plantations, as well as the mechanisation and intensification of the cropping system (De Graaff & Eppink, 1999; Beaufoy, 2003). The study of Beaufoy (2003) concludes that the production support of the CAP during the 1990s has rewarded intensification and expansion and has produced negative environmental effects, of which soil erosion is considered as the most serious environmental problem.

In traditional orchards, the SWC structures were aimed to capture or prevent runoff in order to increase the water availability to the trees (Ennabli, 1993). This is no longer the case in modern orchards, where the trees are planted on a regular grid. Although this does not prevent runoff losses, it maximises the soil volume that can be mined for water by the roots of each tree. In locations with additional water sources, it is common to apply extra water by means of drip irrigation (Beaufoy, 2003). The accelerated soil erosion in modern plantations is of little concern to farmers, in places where the orchards are located on soft rocks like marls, shales and slates. The corresponding soils are marginal, have a low organic matter content, and can be easily ripped by tillage to provide a substrate for the trees (Faulkner et al., 2003; Oñate & Peco, 2005; Ramos & Martínez-Casasnovas, 2006). It therefore appears that the high erosion rates do not result in a reduction in the orchard productivity over the long term. Nevertheless, the erosion processes produce a number of undesirable off-site effects, like for example the siltation of reservoirs (Oostwoud Wijdenes et al., 2000). The challenge is thus to develop new approaches to minimise the negative effects of accelerated soil erosion in tree plantations, that are acceptable to the different stakeholders involved. Examples of such approaches are no-tillage techniques and the application of vegetation strips and cover crops. The advantage of these measures is that they do not hinder mechanisation as they are integrated in the cropping system. Recent studies show that these measures can be economically feasible when properly managed and adapted to local conditions (Gómez et al., 1999; Pastor, 2004; Hernández et al., 2005; Mygdakos et al., 2005; Bueno et al., 2007). A potential drawback of cover crops in dry areas is that they may increase the water requirement of the cropping system and reduce tree vigour and yield (Ingels et al., 1998; Hartwig & Ammon, 2002). Few studies have addressed this topic in rainfed orchards, so that little is known about the conditions and physical thresholds for the successful implementation of cover crops.

1.3 Objectives

This thesis addresses the dilemma of optimising water availability and limiting soil degradation in extensive rainfed orchards on dryland marginal soils. The research objectives are to:

1. Quantify the water balance of the topsoil, in order to evaluate the benefits and drawbacks of the current practice of frequent tillage in terms of soil evaporation and water availability for the tree crop.
2. Investigate to what extent cropping systems of rainfed orchards are adapted to spatial patterns of available water determined by lithology (regional scale) and climate (Mediterranean scale), and how these patterns influence the feasibility of cover crops.
3. Determine the effect of the prolonged decline of terraces and other traditional SWC measures on hydrological connectivity and peak discharge at the catchment scale.
4. Compare the water balance and productivity between an olive cropping system with regular tillage and one that incorporates a cover crop.

1.4 Outline of the thesis

Chapter 2 describes some aspects of the major rainfed cropping systems in areas with a Mediterranean climate, as well as the calculation of crop water requirements and the concept of water use efficiency. Special attention is paid to land management and the causes of land degradation. Subsequently, the recent history of agricultural land use in the northern Mediterranean (Spain, Italy, Greece and Portugal) is discussed, including the role of climate change and European policy. The last part of Chapter 2 is dedicated to possible solutions for land degradation, with specific attention to vegetation measures.

Chapter 3 consists of a detailed analysis of the water balance of rainfed almond orchards at an annual rainfall of 300 mm. It is investigated if the current practice of frequent tillage limits the water availability for the trees, and if there are opportunities to decrease the evaporation from the bare soil. The study includes the quantification of overland flow at the field scale and its effect on the spatial variability in tree vigour. Furthermore, the water balance study will clarify if there are any opportunities for vegetation measures in this dry climate.

Chapter 4 deals with patterns of water availability related to differences in lithology (soil type) and climate. The vulnerability of the orchards and the eventual surplus water for cover crops in rainfed cropping systems strongly depends on the amount of available water for the tree crop (Ingels et al., 1998). Knowledge on patterns of water availability will contribute to the identification of conditions and thresholds for the application of vegetation measures.

Chapter 5 addresses the effect of the prolonged decline in terraces on hydrological connectivity and peak discharge. The study uses a quantitative measure of connectivity that was first introduced by Allard et al. (1994) and Western et al. (1999). The degree of connectivity is determined for different stages of terrace removal and failure, and as a function of storm magnitude. Furthermore, attention is paid to the role of different types of land use on the production of concentrated overland flow and the development of connections between land units.

Chapter 6 reports a field experiment in an olive grove near Sevilla, Spain, where the annual rainfall is about 520 mm. The orchard water balance is compared between a cover crop treatment and a management consisting of regular tillage. This is done by means of soil moisture measurements and a water balance model. Furthermore, the model is used to study the effect of the length of the cover crop growing season on evaporation and soil moisture.

The major findings of the thesis are compiled in Chapter 7, followed by a review of topics that deserve further research.

CHAPTER 2
OVERVIEW OF RAINFED CROPPING SYSTEMS IN AREAS WITH A MEDITERRANEAN CLIMATE

The content of this chapter is based on: European Commission, 2007. Conditions for restoration & mitigation of desertified areas using vegetation (RECONDES) - Review of literature and present knowledge. European Commission, Brussels. ISBN: 92-79-03072-8.

2.1 Introduction

This chapter consists of a literature review on rainfed cropping systems in areas with a Mediterranean climate. The aim is to provide a general background of the subjects addressed in this thesis. The text is primarily focused on the northern Mediterranean (Spain, Italy, Greece and Portugal), but also includes some information from California in the United States of America (USA) and other Mediterranean countries.

2.2 Cropping systems and land management

2.2.1 Crop types and cropping systems

In the northern Mediterranean, the most important crops grown under rainfed conditions are olive, grapevine, almond, barley and wheat. This section lists some properties of these crops and corresponding cropping systems. Furthermore, mixed cropping systems that combine two or more crops are illustrated. Other crops grown under rainfed conditions include sunflower, oats and lentils (Caballero, 2001). Information on the water requirements of cereals, olive, almond and grapevine is provided in §2.2.2 and §2.2.4.

Olive (*Olea europaea* L.)

The olive is an evergreen tree with a biennial growth cycle that results in one main harvest every two years (Ferguson et al., 1994, p. 55; Rallo & Cuevas, 2004; Tubeileh et al., 2004). The extent of this alternate bearing is greater when tree vigour is low, for example as a result of water and nutrient stress (Ferguson et al., 1994, p. 55). The tree density in modern irrigated plantations is higher than in traditional systems, because water availability in the latter is lower and trees need to extract water from a larger soil volume to be able to meet their water requirement (Beaufoy, 2003). Biological threats to olive production are the olive fly (*Bactrocera oleae* Gmelin), other insects and some bacteria and fungi. Detailed information on the nature and management of pests and diseases is provided by Ferguson et al. (1994, p. 97-111), Alvarado et al. (2004), Trapero & Blanco (2004) and Tubeileh et al. (2004). In the northern Mediterranean, 3-6% of the olive production is used for direct consumption (table olives) and the rest is processed to oil (Beaufoy, 2003). A wide range of olive cropping systems is present. According to the level of inputs and management, Beaufoy (2003) distinguishes three broad types of plantation: traditional, intensified traditional and modern. These are described as follows (cited from Beaufoy, 2003):

- Low-input traditional plantations and scattered trees, often with ancient trees and typically planted on terraces, which are managed with few or no chemical inputs, but with a high labour input.

- Intensified traditional plantations which to some extent follow traditional patterns but are under more intensive management making systematic use of artificial fertilisers and pesticides and with more intensive weed control and soil management. There is a tendency to intensify further by means of irrigation, increased tree density and mechanical harvesting.
- Intensive modern plantations of smaller tree varieties, planted at high densities and managed under an intensive and highly mechanised system, usually with irrigation. A second type of modern plantation uses “bushes” in dense rows, resembling vineyards, which are almost totally mechanised.

Almond (*Prunus dulcis* Miller)

In contrast to the olive, the almond tree is deciduous and does not feature alternate bearing. The major commercial cultivars are self-unfruitful and need cross-pollination to produce almonds (Micke, 1996, p. 98-102). Therefore, in almond plantations it is common to alternate one row of a major cultivar with one row of a pollinizer cultivar (Micke, 1996, p. 98-102). Edstrom & Micke (1998), found that significantly higher yields can be reached by alternating two cultivars within rows, emphasising the importance of tree configuration. An overview of pests and diseases is provided in Micke (1996).

Grapevine (*Vitis vinifera* L.)

A study from Roujan, southern France, suggests that the variation in cultivation practices in vineyards consists mainly of differences in field size, tillage and weeding method (Leonard & Andrieux, 1998). Field sizes range from 0.1-2 ha and weeding is done during the entire growing season, from January to July. A common weeding method consists of shallow tillage between the grapevine rows combined with chemical weeding of the rows themselves. An alternative is total chemical weeding in stead of tillage (Leonard & Andrieux, 1998). The soil surface of the vineyards is covered by a varying amount of weeds, depending on the timing of the tillage and chemical weeding. In the Roujan catchment, grasses covered up to 96% of the surface of fields under no-till management, depending on soil moisture content (Leonard & Andrieux, 1998).

Cereals

Both winter barley (*Hordeum vulgare* L.) and winter wheat (*Triticum aestivum* L.) are cultivated in the northern Mediterranean. They are sown in autumn and harvested before the start of the dry summer. Barley can be grown at drier locations than wheat because it matures 2-3 weeks earlier (Metochis & Orphanos, 1997). In this way, barley is less affected by droughts at the end of the growing season and produces higher and more stable yields (Metochis & Orphanos, 1997).

Cover crops

Cover crops are defined here as crops that are grown to provide soil cover during winter and fallow periods in annual cropping systems and crops that are grown next to the main crop to provide soil cover in perennial cropping systems. Cover crops are often grown in cereal based cropping systems such as cereal-legume instead of cereal-fallow rotations (Russi & Falcanelli, 1999; Christiansen et al., 2000; Díaz-Ambrona & Míngues, 2001; Denison et al., 2004). Furthermore, literature is available on the cultivation of leguminous forage crops like clover (*Trifolium incarnatum* L.) in the Mediterranean (e.g. Russi & Falcanelli, 1999). The application of cover crops is still rare in the Mediterranean. Díaz-Ambrona & Míngues (2001) report that grain legumes cover less than 3% of the cultivated land in Central Spain. Legumes that are cultivated there are faba bean (*Vicia faba* L.), pea (*Pivum sativum* L.) and lupin (*Lupinus albus* L.). In California, US, 16% of the grapevine area is sown with cover crops (Ingels et al., 1998). Out of eleven examples of vineyards with cover crops two involved rainfed conditions. Furthermore, Ingels et al. (1998) provide information on a broad selection of cover crops, including both annual and perennial legumes and grasses. A distinction is made between species for tilled and no-till vineyards. The application of a mixture of cover crops may have advantages compared to one single crop (Ingels et al., 1998). For example, if the vineyard is heterogeneous with regard to available soil moisture, one species may grow better in the drier areas and the other in the wetter areas, maximising the soil cover throughout the vineyard. Research by Pastor (2004) demonstrated that it is possible to grow cover crops in rainfed olive plantations during winter. The cover crop consisted of weeds located in between the rows of trees, whereas the soil beneath the trees was kept bare by chemical weeding. When this cover crop is present only during the winter months and is removed in spring, its water use is relatively low (Pastor, 2004). The most widely used cover crops in almond plantations in California are vetch, Blando bromegrass and clovers (Micke, 1996, p. 196-201). Resident vegetation might be a good cover crop as well and involves less management and lower overall costs (Micke, 1996, p. 196-201). A list of cover crop species for use in rainfed cropping systems is provided in Appendix 1.

Mixed cropping systems

Some examples of mixed cropping systems are given here. These are traditional cropping systems, that combine the cultivation of cereals and perennial crops. In modern agriculture, these systems are becoming rare. Instead, new agricultural fields are large and aimed at the production of a single crop. Examples of the intensification and the economy of scale are the modern olive and almond plantations in Spain and the practice of land levelling in Tuscany, Italy (Clarke & Rendell, 2000; Martínez-Casanovas & Sánchez-Bosch, 2000; Borselli et al., 2002; Ramos & Mulligan, 2005). Traditional mixed cropping systems are the dehesa system in West Spain, the montado system in South Portugal and the coltura mista in Italy (Joffre et al., 1988; Ceballos & Schnabel, 1998; Pinto-Correia & Mascarenhas, 1999). The montado features a widely spaced tree stand, some 20-80 trees ha^{-1} and a ground cover of arable crops like barley and wheat in rotation with fallow and pasture (Pinto-Correia & Mascarenhas, 1999). Typical trees in such systems are cork oak (*Quercus suber* L.), holm oak (*Quercus rotundifolia* L.), some mountain oaks (*Quercus pyrenaica* Willd.), olive and sweet chestnut (*Castanea sativa* Miller; Pinto-Correia & Mascarenhas, 1999). The coltura mista combines the cultivation of olive, grapevine and cereals. The difference with the montado is that coltura mista involves no pasture and livestock and that the climate is more humid.

2.2.2 Crop water requirements

Crop water requirement can be defined as "The depth of water needed to meet the water loss through evapotranspiration of a disease-free crop, growing in large fields under non-restricting soil conditions including soil water and fertility, and achieving full production potential under the given growing environment" (Allen et al., 1998). While evapotranspiration is difficult to measure (Allen et al., 1998), a great amount of research has been done on the measurement and estimation of crop evapotranspiration since the 1950's. An overview of methods to measure and estimate crop evapotranspiration is provided by Rana & Katerji (2000). These include: soil water balance, weighing lysimeter, energy balance and Bowen ratio, aerodynamic method, eddy covariance, sap flow method, chamber system and Penman-Monteith method. The Penman-Monteith method is a method to calculate evapotranspiration from climatic parameters and was adapted to calculate crop water requirements by Doorenbos & Pruitt (1977). In 1990, the Food and Agricultural Organisation (FAO) organised a consultation of experts and researchers to advise on the revision of this method (Allen et al., 1998). The revised method was named the FAO Penman-Monteith method and is documented by Allen et al. (1998). It is the most widely applied method for the calculation of crop water requirement and irrigation planning in agriculture.

In literature, the crop evapotranspiration under optimal conditions is referred to as the crop evapotranspiration under standard conditions (Allen et al., 1998), the maximum crop evapotranspiration or the potential crop evapotranspiration (Metochis & Orphanos, 1997), mostly denoted by ET_c. We will use here the term potential crop evapotranspiration and notation ET_c for the evapotranspiration under optimal conditions. Subsequently, we distinguish ET_c from evapotranspiration under non-optimal conditions, or actual crop evapotranspiration, denoted by ET_a. The FAO Penman-Monteith method uses crop coefficients and a reference crop evapotranspiration (ET_0). ET_0 is defined as the evapotranspiration of a hypothetical reference crop with an assumed crop height of 0.12 m, a fixed surface resistance of 70 s m^{-1} and an albedo of 0.23 (Allen et al., 1998). ET_0 is expressed in units of water depth per unit of time, e.g. mm day^{-1}. ET_c is then calculated by multiplying ET_0 with a crop coefficient (K_c). Hence, the crop coefficient relates ET_c to ET_0 for a disease free crop grown under optimal conditions in terms of water and soil fertility (Allen et al., 1998). The K_c value that should be used depends amongst others on the crop type, crop growth stage, management and environmental conditions. It can be determined by measuring ET_c in the field.

The most widely used methods to compute ET_0 are the FAO Penman-Monteith equation, the pan evaporation method and the Hargreaves equation (Allen et al., 1998). Because ET_0 is only a function of climatic parameters, it can be computed from weather data (Allen et al., 1998). The Penman-Monteith equation requires maximum and minimum temperature, relative humidity, wind speed and sunshine duration or radiation. Missing humidity, radiation and wind speed data can be estimated (Allen et al., 1998). An alternative to calculate ET_0 is the pan evaporation method, which does not need recordings of climate variables. With the evaporation pan the daily evaporation from an open water surface is measured, which is converted to an estimation of ET_0 (Allen et al., 1998). The pan should be established at a site that closely resembles the area of study with regard to climate. Detailed information on theory and application of these methods is provided by Allen et al. (1998). The Hargreaves equation forms more simple method to calculate ET_0 and requires only temperature data (Hargreaves & Samani, 1985). Although it is less accurate, the Hargreaves equation produces reasonable estimates of ET_0 for longer time intervals like weeks and months. It tends to underestimate ET_0 when wind speeds are high and to overestimate ET_0 under conditions of high relative humidity (Allen et al., 1998). The equation has been shown to be robust in Mediterranean environments with low wind speeds (Orgaz & Fereres, 2004).

2.2.3 Water use efficiency and water harvesting

The water use efficiency (*WUE*) of a cropping system can be defined as the amount of biomass, dry matter or harvestable product that is produced per unit of water applied (kg mm^{-1} ha^{-1}; Lövenstein et al., 1995; López & Arrúe, 1997; Cantero-Martínez et al., 2003). At the plant level, one can define a similar parameter called the transpiration efficiency. This is the mass of dry matter produced per unit of water transpired (Lövenstein et al., 1995). The *WUE* is mostly used to indicate and compare the performance of cropping systems in terms of water use for the entire growing season (e.g. López & Arrúe, 1997; Cantero-Martínez et al., 2003). In the case of irrigated crops the *WUE* incorporates the efficiency of the irrigation method used. In some cases, a different management can increase the *WUE* of a cropping system. An example is the application of drip- instead of flood-irrigation (e.g. Beede & Goldhamer, 1994). For rainfed cropping systems, an increased *WUE* may enable successful cultivation under drier climatic conditions. A broader picture on *WUE* is provided by Wallace (2000), who indicates the importance of *WUE* for world food production in the future and considers the potential to improve *WUE*.

Water harvesting methods aim to maximise *WUE* under rainfed conditions. There is a great variety of traditional and modern water harvesting methods. Some techniques are based on the direct redistribution of runoff water during storms. For example in Tunisia, one traditional method on sloping fields is to leave the upper part of the slope bare and to cultivate crops on the lower part. The runoff generated on the upper part is directly routed to the crop on the lower slope, and retained by small dikes (Ennabli, 1993). The routing of runoff to the crop is a common element of many traditional techniques (Ennabli, 1993). An alternative is to collect runoff in tanks, cellars and reservoirs (Ennabli, 1993; Van Wesemael et al., 1998; Yuan et al., 2003). The advantage of storage is that the application of the water can be timed according to the specific requirements of the crop. In some traditional cropping systems, trees are deliberately planted at points of runoff concentration, for example at the foot of the slope or at the upslope side of bench terraces and retention ditches. This occurs for example in south-east Spain, as illustrated on an aerial photo in Van Wesemael et al. (2003).

For perennial crops, a method to maximise production is to adapt the crop spacing to the local climate (Tubeileh et al., 2004). The roots of trees like almond and olive are able to mine the soil around their trunk. If the climate is drier and water availability lower, individual trees need a greater volume of soil to meet their water requirement. Hence, at sites with lower water availability, the trees need to be spaced further apart for optimal production. This explains the rationale behind traditional tree arrangements, with up to 24 m of bare soil between rows of trees, as still present in Tunisia (Ennabli, 1993). The relation between climate and tree spacing is addressed in more detail in Chapter 4.

When cover crops are applied, the optimal tree configuration may be different and may have to be adapted. If cover crops prove to use a part of the water that had otherwise been available to the trees, the tree density will have to be lowered to remain optimal. This could be studied by quantifying the components of the water balance of the cropping system.

Another way to increase water availability and water use efficiency is to apply a mulch. The aim of the mulch is to reduce the soil evaporation. There are many materials that can be used for mulching, including crop residues, straw (Mellouli et al., 2000), stones (Groenevelt et al., 1989; Gale et al., 1993; Li, 2003) and olive mill effluent (Mellouli et al., 2000). The effect of mulching on crop growth conditions and production is illustrated here for gravel-sand mulches. The labour requirements to establish a gravel-sand mulch are relatively high, but it may last up to 60 years (Li, 2003). The elaborate study of Li (2003), showed the following effects of the mulch on conditions for plant growth compared to the bare soil (> 65% silt): the soil moisture between 20 and 60 cm depth was higher; runoff was lower; evaporation was lower, soil temperature fluctuations were smaller; soil temperature was up to 5 °C higher, and advantage in areas prone to early-season frost; great reduction of wind erosion; higher activity of micro organisms. The magnitude of these effects decreased with the age of the mulch. Accordingly, the grain yield of spring wheat was 2500, 1700 and 1149 kg ha^{-1} for respectively 3, 9 and 50 year old fields. The yield on a field without mulch was 600 kg ha^{-1} (Li, 2003).

2.2.4 Water requirement and water use efficiency of some crops in the Mediterranean

Winter barley (*Hordeum vulgare* L.) and wheat (*Triticum aestivum* L.) are the most widely grown cereals in the Mediterranean region. Wheat occurs in the northern parts, whereas barely dominates the drier areas with an annual rainfall of less than about 500 mm (López & Arrúe, 1997; Metochis & Orphanos, 1997; Cantero-Martínez et al., 2003). In Syria, the water requirement throughout the growing season of four bread wheat cultivars was reported to range from 390 to 485 mm for optimal production, whereas the measured ET_a under rainfed conditions was 246 to 328 mm (Oweis et al., 2000). In the central plain of Cyprus a 13-year experiment was conducted on barley by Metochis & Orphanos (1997). They observed a water requirement of about 240 mm for optimal crop growth. For the estimation of ET_c, K_c values for barley and wheat are provided by e.g. Allen et al. (1998) and Metochis & Orphanos (1997).

In northern Spain, barley is traditionally grown in a barley-fallow rotation in areas with less than 500 mm of annual rainfall. At higher rainfall rates annual cropping is practised (López & Arrúe, 1997). In the barley-fallow rotation, the soil is kept bare from weeds during fallow by regular tillage. The aim of the fallow year is to reach an as high as possible soil moisture content at the start of the next growing season. Austin et al. (1998) estimated that in northern Spain the yield of barley in rotation with a fallow year is 15% higher compared to rotation with another crop. Nevertheless, they conclude that despite the higher yield, the barley-fallow rotation is uneconomic compared to annual cropping. Furthermore, López & Arrúe (1997) stated that a fallow year is an inefficient practice for improving soil water storage and subsequent crop yield. Lampurlanés et al. (2002) studied the effects of different tillage regimes on the soil water content during fallow. The subsoil, minimum and no-tillage treatments did not produce significant differences in soil water storage. Results of Agenbag & Maree (1991) show a somewhat higher soil water content for no-tillage compared to mouldboard and tine tillage. This did not affect barley yields, except for dry years, where yields where higher for the no-tillage treatment (Agenbag & Maree, 1991). For example López et al. (2003), argue that no-tillage is the preferable management option to minimise soil loss by wind and water erosion. It provides soil cover by crop residues outside the growing season (López et al., 2003).

When considering cereal cultivation in semi-arid areas, some studies highlight the influence of nitrogen fertilisation and tillage practice on the *WUE*. López & Arrúe (1997) found that *WUE* and yields are significantly lower in no-tillage systems compared to conventional and reduced tillage. Therefore, only reduced tillage would be an alternative to conventional tillage, maintaining yields at an acceptable level. Both studies on barley and wheat show a positive relation between N-fertilisation and *WUE* (Oweis et al., 2000; Cantero-Martínez et al., 2003). This is directly related to the increased grain yield with N-application. Simpson & Siddique (1994) studied the effect of soil texture on yield and *WUE* of barley and wheat in western Australia. They observed average *WUE* values based on grain yield of 11 kg ha^{-1} mm^{-1} for barley and 9 kg ha^{-1} mm^{-1} for wheat. The cumulative ET_a for the entire growing season was respectively 236 and 253 mm (Simpson & Siddique, 1994). In the Ebro valley in the northeast of Spain, Cantero-Martínez et al. (2003) compared the *WUE* of barley on three sites with different climate. They observed a *WUE* based on grain yield of 5 kg ha^{-1} mm^{-1} on the driest site (annual rainfall 352 mm) and 11 kg ha^{-1} mm^{-1} on the wetter site (annual rainfall 475 mm).

Olive, almond and grapevine

For perennial crops the ET_c and K_c are often site specific and generalisations are more difficult than for annual crops. For different combinations of crop cover and age, crop spacing, presence of bare soil, presence of cover crops and specific crop management, ET_c and the K_c value will be different (e.g. Tubeileh et al., 2004). In the Mediterranean, there is a large variety in cropping systems (Beaufoy, 2003; Navarro & Parra, 2004). One approach to estimate ET_c for different situations is to adjust the K_c based on site-specific information (Allen et al., 1998). For example, ET_c can be multiplied by a reduction coefficient (K_r), which represents the effect of the fraction of the orchard floor covered by crop canopy (Palomo et al., 2002; Orgaz & Fereres, 2004). In this case, ET_c is related to ET_0 by both K_c and K_r (Palomo et al., 2002; Orgaz & Fereres, 2004). It should be noted that the K_c used in combination with K_r is not the same as the K_c of the approach of Allen et al. (1998) and will have different values. Some alternative, model-based methods to determine the actual crop evapotranspiration of perennial crops are presented by Sene (1996), Trambouze & Voltz (2001) and Villalobos et al. (2000). They require the measurement of parameters that are not readily available from weather stations. Still other parameters have to be estimated and the models require calibration.

In general, for perennial crops a limited amount of knowledge exists on the effect of various tillage regimes on water availability and yield. Gómez et al. (1999) observed no significant overall difference in olive yield with no-tillage compared to conventional tillage. During the 15-year study in Cordoba, Spain, only in one very dry year a significantly higher yield was obtained for the no-tillage treatment (Gómez et al., 1999).

An overview is now given of data related to the water requirement of olive, almond and grapevine that was encountered in literature. For olive, information on crop coefficients is scarce (Villalobos et al., 2000; Palomo et al., 2002). Nevertheless, in total nine sources were encountered that provide figures of K_c and or evapotranspiration. K_c values reported for the K_c & K_r approach range from 0.45 to 0.75 (Orgaz & Fereres, 2004; Tubeileh et al., 2004). These figures are from orchards with a bare soil. It should be noted that the K_c changes during the year, being relatively high in spring and autumn and low in summer (Orgaz & Fereres, 2004). When using K_c only, the reported K_c values range from 0.55 to 0.75 for orchards without cover crops (Beede & Goldhamer, 1994; Allen et al., 1998; Villalobos et al., 2000). For orchards with cover crops, Beede & Goldhamer (1994) list values between 0.95 and 1.05. Figures of ET_c and ET_a were encountered in six sources (Beede & Goldhamer, 1994; Villalobos et al., 2000; Palomo et al., 2002; Moriana et al., 2003; Nuberg & Yunusa, 2003; Tubeileh et al., 2004). Villalobos et al. (2000) reported an ET_c of 855 and 758 mm a^{-1} for respectively an intensive and a traditional olive plantation in Cordoba, Spain. For rainfed orchards in south-west Spain, at an ET_0 of 1420 mm a^{-1}, Moriana et al. (2003) observed an ET_a of 508 mm a^{-1} and Palomo et al. (2002) an ET_a of 378 mm a^{-1}.

For almond, Goldhamer (1996) provides K_c and ET_c values for the San Joaquin Valley, California, US. They are valid for orchards with a tree spacing of 6.7 by 7.3 m with a bare soil. The long term average ET_c in the area is 991 mm during the almond growing season, which is from 15 March to 16 November. The K_c value increases from 0.54 in March to 0.94 in July and August. From September to November it decreases to 0.70. This change throughout the year is related to the growth and senescence of the leaves (Goldhamer, 1996). Romero et al. (2004) used K_c values of 0.22 to 0.61 in a three-year experiment in Murcia, Spain, with a tree spacing of 7 by 5 m. The plantation probably had a bare soil, although the authors do not explicitly mention that, and yearly ET_0 ranged between 1099 and 1281 mm. Allen et al. (1998) list K_c values from 0.40 to 0.90 for almond plantations with a bare soil, depending on crop growth stage within the year. In Sfax, Tunisia, Ghrab et al. (2002) compared the performance and yield of 24 almond cultivars in rainfed conditions during a 5-year experiment with an average annual rainfall of 250 mm.

Due to time constraints, less information was gathered on grapevine than on olive and almond. Accordingly, the data provided here only give an impression and do not incorporate all recent literature on the water requirement of grapevine. Allen et al. (1998), list K_c values of 0.30 to 0.70 for wine grapevine and 0.30 to 0.85 for table grapevine. Trambouze & Voltz (2001) measured the daily transpiration rate of a grapevine crop in Southern France, from about 3 mm day^{-1} in July to 1 mm day^{-1} in August (Fig. 2.1).

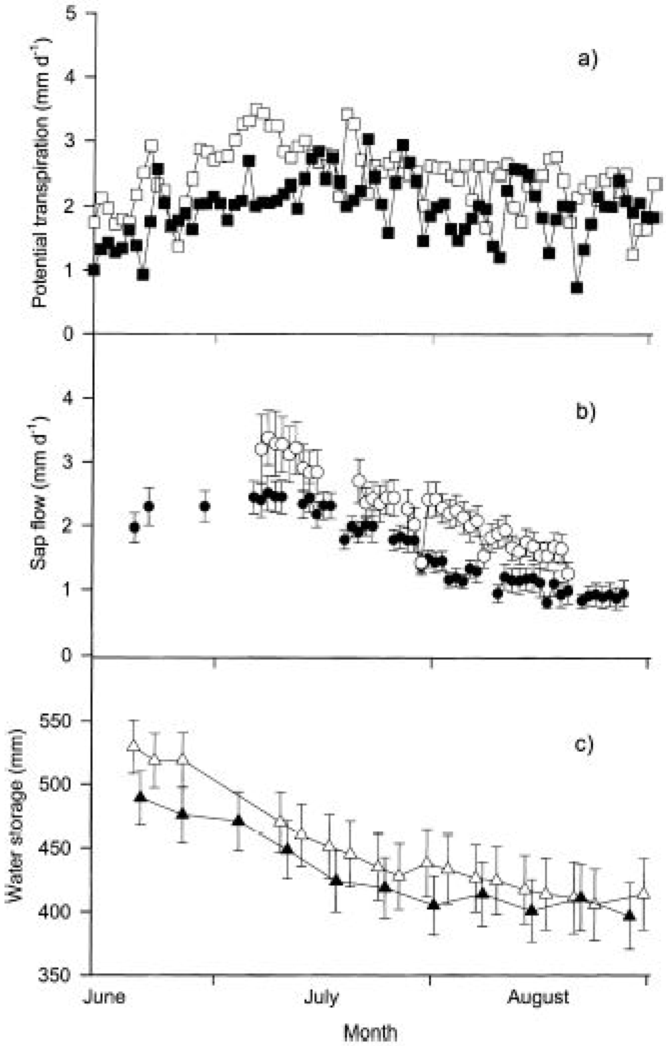

Figure 2.1. Field measurements of (a) potential transpiration, (b) actual field transpiration and (c) soil water storage of a grapevine crop in south France, during the summers of 1994 (open symbols) and 1995 (closed symbols). The vertical bars represent 95% confidence intervals. Source: Trambouze & Voltz, 2001.

The literature on perennial cropping systems does not provide much information on their *WUE*. A reason might be that *WUE* values based on only a few years of harvest data are not considered very useful because of the large yield fluctuations between years. For almond, three sources did report yield data that could easily be converted to *WUE* figures. Two year of harvest data of a study by Romero et al. (2004) in Murcia, Spain, show a *WUE* based on kernel yield of 1.58 kg ha^{-1} mm^{-1} for a fully irrigated almond grove with 13-year-old trees. In another treatment, they applied a deficit irrigation equal to 20% of the calculated ET_c during the kernel-filling period from June to August. The *WUE* of this treatment was 1.71 kg ha^{-1} mm^{-1}. Godini & Palasciano (1998) studied the growth of ten almond cultivars in a 13-year trial near Bari, Italy. The trees were grown under rainfed conditions and were 5 years old at the start of the experiment. The average *WUE* based on kernel yield of all cultivars from 1984 to 1994 was 0.47 kg ha^{-1} mm^{-1}. The highest yielding cultivar was Genco, with a *WUE* of 0.71 kg ha^{-1} $mm^{-1}$and the lowest yielding was Nonpareil with a *WUE* of 0.22 kg ha^{-1} mm^{-1}. Yield data from Ghrab et al. (2002) could not be converted to per hectare *WUE* figures. The experiment in Sfax, Tunisia involved 24 cultivars grown under rainfed conditions. Average annual yields based on five years ranged from 3.8 kg $tree^{-1}$ for Tunisian cultivars to 9.7 kg $tree^{-1}$ for Spanish and Italian cultivars.

Some information on methods to increase the *WUE* of perennial crops is provided by Tubeileh et al. (2004) and Romero et al. (2004). For example, regulated deficit irrigation may increase the *WUE* in tree plantations in marginal rainfed conditions (Romero et al., 2004; Tubeileh et al., 2004).

2.2.5 Land management

The land and soil management on rainfed croplands is determined by individual farmers and land owners (Beaufoy, 2003). Two forms of land management can be distinguished: annual soil management and reshaping of the land, e.g. construction of terraces, land levelling. The annual management of cropping systems depends on crop type, soil type and weather (Beaufoy, 2003). The perennial cropping systems of olive, almond and grapevine may involve tillage of the soil. The aim is to prevent weeds to compete with the main crop and to minimise water loss through transpiration by weeds and soil evaporation (Poesen et al., 1997; De Luna et al., 2000; Beaufoy, 2003; Tubeileh et al., 2004; Pastor, 2004). Tillage tools mentioned in literature are cultivator, tine-harrow, disk-harrow, rotovator, duckfoot chisel and mouldboard plough (Poesen et al., 1997; De Graaff & Eppink, 1999; Beaufoy, 2003; Pastor, 2004). The number of tillage passes per year varies from one to two in extensive orchards to five in intensive systems (Poesen et al., 1997; Beaufoy, 2003). In the case of high rainfall, more weeds grow and more tillage is required to keep the soil bare (Beaufoy, 2003). In literature, contour tillage is preferred to tillage up and down the slope. The latter is still practiced, especially in vineyards (Poesen et al., 1997). The control of weeds without tillage is possible with herbicides, which are used since the 1960s (De Graaff & Eppink, 1999; Gómez et al., 1999; Beaufoy, 2003). Additional roles of tillage in annual cropping systems are the preparation of the seedbed and the incorporation of crop residues after harvest (Díaz-Ambrona & Mínguez, 2001). During a fallow year, weeds may be cleared two to four times by harrowing, as reported for the Ebro valley in Spain (Lampurlanés et al., 2002). The equipment mentioned in literature with respect to annual cropping systems consists of mouldboard plough, chisel plough, disc plough, tined cultivator, disc harrow and spike-toothed harrow (Pala et al., 2000; Mas & Verdú, 2003). Land management may also consist of actual reshaping of the land, e.g. the levelling of slopes, creation of terraces or the clearance of land for new tree plantations. At present, there is a trend in Italy and Spain to increase the size of agricultural fields (Clarke & Rendell, 2000; Martínez-Casanovas & Sánchez-Bosch, 2000; Borselli et al., 2002; Torri et al., 2002; Ramos & Mulligan, 2005). Slopes are levelled and terraces removed by bulldozer. This has large consequences in terms of soil properties and erosion.

2.2.6 Land degradation processes

The land degradation associated with dryland agriculture in the Mediterranean is directly related to land management. When the soil is kept bare, rain events may result in considerable soil erosion. This is the case when tillage and land levelling are practiced (e.g. Borselli et al., 2002; Torri et al., 2002; Borselli et al., 2006; Van Oost & Govers, 2006). Typically, bare soils do occur in olive and almond plantations where clean sweeping (i.e. frequent shallow tillage) is practiced to remove weeds, during fallow periods between harvest and sowing of annual crops and during fallow years that involve clean sweeping. The process that results in soil redistribution on slopes is referred to as tillage erosion (Revel et al., 1989; Lindstrom, 1992; Poesen et al., 1997; Van Oost & Govers, 2006). Kinetic sieving induced by tillage results in the concentration of stones at the soil surface, and a distribution of rock fragment cover that is strongly related to topography (Poesen et al., 1997; Oostwoud Wijdenes et al., 1997). Rock fragments and fine earth are transported from the convex hilltops over the slopes and deposited at the edges of concavities. The floor of narrow valleys becomes completely covered by these stony deposits. Such tillage-induced patterns of rock fragment cover and soil depth have specific repercussions in terms of hydrologic and degradation processes (Poesen et al., 1997) as well as plant and tree growth (Van Wesemael et al., 2000). Poesen et al. (1997) demonstrate that the soil loss by frequent contour tillage of stony soils can amount to 31 Mg ha^{-1} a^{-1} for a field on a 50 m long slope of 20% (Poesen et al., 1997). De Alba (2003) performed a model study for a field with a 54 m long slope and gradients between 17 and 40%. The result showed a net soil loss on 59% of the slope surface at an average rate of 16.7 Mg ha^{-1} for one contour tillage pass. In terms of cropping systems, the highest erosion rates are associated with olive and almond plantations and vineyards (Beaufoy, 2003; Martínez-Casanovas & Sánchez-Bosch, 2000). For instance Faulkner et al. (2003) illustrate the erosion risk associated with large scale clearance of soils for tree plantations in Almeria, Spain. Oostwoud Wijdenes et al. (2000) found that the most active gullies in south-east Spain are incised in almond and wheat fields on marls. They hypothesised that this is due to higher runoff and soil erodibility on cultivated fields. Another process associated with soil erosion in abandoned fields is the collapse of terrace walls (Ruecker et al., 1998).

2.3 Agricultural land use since 1960

2.3.1 Extent and crop yields

From 1961 to 2001 there has been an overall decrease in agricultural land from 66 to 57% in Spain, Italy, Greece and Portugal. The area of arable land decreased by 25% whereas the area of perennial crops increased with 11% (FAOSTAT data, 2004). For the period 1930-1993, Debussche et al. (1999) observed an increase in woodland cover and tree height and a decrease in cropland and rangeland in southern France. Likewise, Taillefumier and Piégay (2003) reported an increase in bush and forest cover at the expense of formerly agricultural and pastoral land in southern France.

In Spain, from 1961 to 2001 the area with cereals decreased by 7% whereas cereal yields increased from about 1 to 3 Mg ha^{-1} a^{-1} (Fig. 2.2). The yields increased gradually over the entire period. Probably this can be explained by the mechanisation and intensification of agriculture during that period as well as the development of more productive cultivars. Compared to northern Europe the cereal yields in Mediterranean countries remain relatively low. According to Olesen & Bindi (2002) this is the result of the limited water availability, heat stress and the short duration of the grain filling-period.

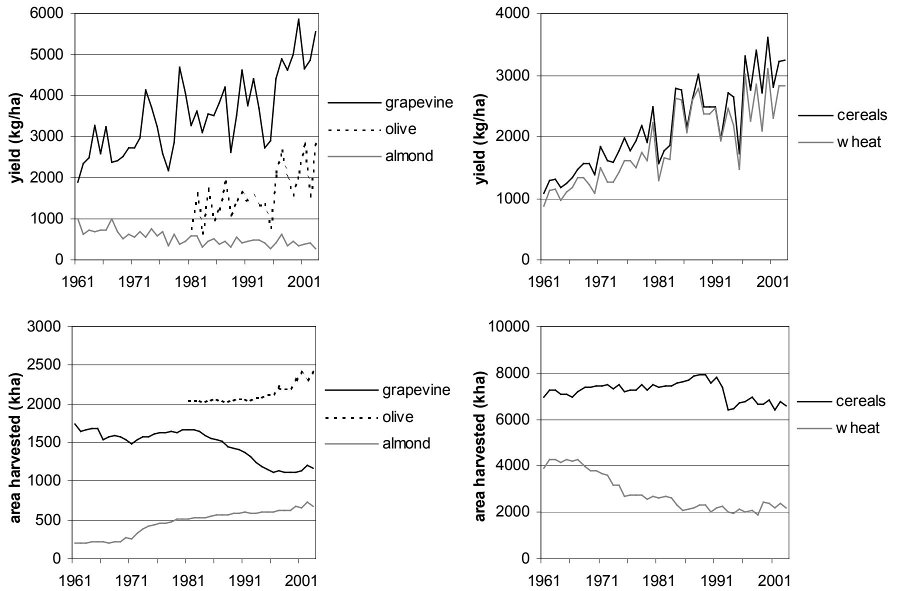

Figure 2.2. Development of yields and harvested area of some crops in Spain (FAOSTAT data, 2004). The term "cereals" stands for the aggregate of all cereals, like wheat, barley, oat, rye, etc. The olive data series are only available from 1981 onwards.

Between 1961 and 2003 olive and grapevine showed yield increases of more than 100% while almond yields decreased by 50% (Fig. 2.2). For explanation of these figures, some of the developments and properties of those cropping systems are discussed below. This involves changes in for example technology, irrigation practices, climate, farm size and policy. The data of Figure 2.2 represent the average for the whole of Spain. It should be noted that there is actually a considerable spatial variability in yields, which stems for example from the gradient in rainfall from north to south.

2.3.2 Intensification

The intensification and mechanisation of agriculture has been an important development since the 1950's. Accordingly, the subject of intensification and extensification has been identified as an important topic for model studies on agricultural land use change (e.g. Lambin et al., 2004). A recent example is the intensification of the olive production in Spain, after it joined the EU in 1986 (De Graaff & Eppink, 1999). The Common Agricultural Policy (CAP) of the European Union (EU) was identified as the major cause for this intensification (De Graaff & Eppink, 1999; Beaufoy, 2003). An example of the mechanisation of Mediterranean agriculture is the practice of land levelling. Heavy machinery is used to create large agricultural fields on unstable slopes and in badlands (Borselli et al., 2002; Borselli et al., 2006). The associated soil loss and off-site effects can be considerable, especially during rains just after the levelling (Borselli et al., 2002).

2.3.3 The role of EU policies

Changes in Mediterranean agriculture have been induced by the CAP (Olesen & Bindi, 2002). Among its goals are "a fair standard of living and income stability for the agricultural community", "a competitive agricultural sector" and "production methods that support environmentally friendly, quality products that the public wants" (EU, 2002). The policy instruments used by the CAP to implement the EU-vision on agriculture consist of subsidies and price control and policy to develop "less-favoured agricultural areas" (De Graaff & Eppink, 1999). Historically, the CAP was introduced to stimulate the agricultural production (EC, 2007). This has been its main purpose up to the 1980s, leading to permanent surpluses of some major farm commodities. Since the 1990s, the CAP subsidies have been reduced. The CAP reform of 1992 decreased intervention prices by one third and introduced area payments, including set-aside schemes (Olesen & Bindi, 2002). Another reform has come into effect since 2005, in which the subsidies are no longer proportional to production, but are awarded in the form of "single farm payments", which are meant to provide some level of income stability to farmers (EC, 2007). Topp & Mitchell (2003) argue that the CAP has a complex impact. A change in CAP has effects at various scales, e.g. the farm, field and landscape level. Also a policy change will result in a redistribution of support to the rural community that in turn will change land use and land use patterns (Topp & Mitchell, 2003).

2.3.4 EU policy and olive production

The effect of the CAP on agriculture and its environmental impact is illustrated here for olive production in Spain. From 1966 to 1998, small farmers producing less than 500 kg of olive oil per year received a support payment based on tree numbers and historic yields in the district (note that Spain joined the EU in 1986). This provision was found to be impractical due to a lack of reliable data in the olive sector and problems of fraud (EC, 2002). The CAP support for olive oil was therefore reformed in 1998, so that the subsidy was directly coupled to production for all producers. Effectively, the EU subsidies during the 1990s amounted to 50% of the actual sales value of olive oil, which is close to the production cost in rainfed plantations (De Graaff & Eppink, 1999). From 1992 to 2003, the olive production in Spain increased about 80% from 3075 to 5650 Gt a^{-1} (FAOSTAT data, 2004). The yield per hectare increased about 50% (Fig. 2.2). This points to a strong intensification of the olive cropping systems. A negative aspect of that intensification is the high soil erosion rate associated with intensified traditional plantations (De Graaff & Eppink, 1999; Beaufoy, 2003), and with the clearance of natural vegetation on dispersive soils for the expansion of olive groves (Faulkner et al., 2003). The frequent tillage practiced can lead to both water and tillage erosion. (e.g. Van Wesemael et al., 2000; Beaufoy, 2003). The major threat of the high erosion rates observed is the loss of topsoil and biological productivity of the land. In Spain, already 20% of the olive plantations are reported to have lost the upper soil horizon (Lopez Cuervo, 1990 cited in Beaufoy, 2003), indicating the unsustainability of specific forms of this land use. It can be concluded that the production support of the CAP has rewarded intensification and expansion of olive oil production and has produced negative environmental effects (Beaufoy, 2003). The effects of the CAP reform of 2005 (§2.3.3) are not yet clear, because the transition period lasts several years.

2.3.5 Effects of climate change

Climate change studies indicate higher temperatures and less and more variable rainfall in the Mediterranean (Olesen & Bindi, 2002). The temperature increase between 1990 and 2090 is projected to be 3-4 °C in the Mediterranean Basin (IPCC, 2007; SRES scenario A1B). Higher temperatures reduce the length of the growing season of annual crops and their yield. Climate models project that the precipitation in the Mediterranean during the summer months (June to August) may decrease by 20% between the year 2000 and 2100 (IPCC, 2007; SRES scenario A1B). For the winter months (December to February), there is a disagreement between climate models with respect to precipitation changes in the northern half of the Mediterranean region. For the southern half (north Africa), the winter precipitation is expected to decrease by 10% or more (IPCC, 2007; SRES scenario A1B).

Higher CO_2 concentrations will increase yields and the water, nitrogen and radiation use efficiency of the crop (Olesen & Bindi, 2002). For wheat the water use efficiency has been found to increase about 50 to 60% and the yield by 28% for a doubling of the current CO_2 concentration (Downing et al., 2000, cited in Olesen & Bindi, 2002). The combined effect of CO_2, temperature and precipitation changes on Mediterranean wheat yields is not clear. Nevertheless, the trend of a decrease in cereal cultivation and increase of perennial crops is expected to continue. The reason is that cereal yields in the Mediterranean are less than 50% of those in Western Europe, reaching about 2.7 Mg ha^{-1} a^{-1} (Olesen & Bindi, 2002). Therefore, in general, the Mediterranean cereal production is not cost efficient enough to be able to compete with other producers on the EU and world market. For example Pinto-Correia (2000) reports a decline in market-oriented cereal production in the traditional montado system in Portugal.

Rosato & Giupponi (2003) discussed the consequences and impacts of climate change in Mediterranean agriculture and considered possible adaptations. They proposed a method for the downscaling of global change models to regional and local land use scenarios, integrating socio-economics in climate change scenarios. Ewert et al. (2005) and Rounsevell et al. (2005) developed future scenarios of European agricultural land use for the years 2020, 2050 and 2080. The scenarios were based on storylines from the IPCC Special Report on Emission Scenarios (SRES) (Nakićenović et al., 2000). The SRES storylines represent sets of different demographic, social, economic, technological and environmental developments. The four scenarios considered by Rounsevell et al. (2005) were coded A1FI, B1, A2 and B1. This is a distinction between more economically (A) and more environmentally and equity (B) orientated futures, as well as more globally (1) and more regionally (2) oriented developments (Nakićenović et al., 2000). The code FI stands for fossil fuel intensive. See Rounsevell et al.

(2005) and Ewert et al. (2005) for detailed information on the scenario development.

Rounsevell et al. (2005) calculated the future distribution of cropland (arable and permanent crops for food and fibre production) and grassland areas with a supply / demand model. With the parameterisation used, cropland and grassland areas decline by as much as 50% from 2000 to 2080. The decrease is considerably larger for the A1FI and A2 scenarios then for B1 and B2. The decline in grassland is even greater, except for the B1 scenario. When considering regional differences within the EU, the countries bordering the Mediterranean show a larger decrease of agricultural land use than central and northern Europe. This is especially true for the A1FI and A2 scenarios, because large parts of these countries belong to the less favoured areas of the EU in terms of agricultural productivity (Rounsevell et al., 2005).

Ewert et al. (2005) estimated the change in crop productivity in the EU for the scenarios outlined above. They applied a simple statistical approach to project an analysis of historic yield trends of wheat (*Triticum eastivum* L.) into the future. The model used included the effect of climate change, increasing CO_2 concentrations and technology development. By 2080, Ewert et al. (2005) estimated a yield increase of 40 to 160% for the different scenarios. The increase was considerably higher for the A1FI and A2 scenario than for B1 and B2.

2.4 Mitigation of land degradation

2.4.1 Policy and guidelines

At present, a European policy on soil protection is being developed. From EU communication COM/2002/0179 (EU, 2002), it appears that the policy on soil protection will be incorporated in the CAP, transport policy and some other related policies. The communication mentions opportunities for soil protection within the CAP, being support for conservation tillage, the maintenance of terraces, integrated crop management, organic farming, lower stocking density and the use of certified compost.

It can be argued that plans to reduce soil erosion should be made locally at the landscape level and should consider complete catchments or sub-catchments in order to be effective. For example in Belgium, the regional policy on soil erosion is aimed at local authorities through subsidies for local erosion control plans (AMINAL, 2002a, 2002b). Another example of mitigating land degradation at the local scale is the cooperation between farmers in Australia, organised in so called land care groups and involving government support (Campbell, 1994).

A brochure of AMINAL (2002c) forms an example of erosion control guidelines. It includes a description of specific erosion control measures as well as their use and application. The brochure presents measures at both the field and landscape level. It seems that the scientific literature is more focussed on the effect of specific conservation measures and provides few general guidelines. One brochure concerning the application of vegetation measures against soil erosion in semi-arid environments is provided by RECONDES (2007).

2.4.2 Mitigation techniques

Land degradation in agricultural areas should be addressed by a change in cultivation practices and agricultural policy. For the mitigation of soil erosion there are several possibilities, including the construction of conservation structures such as terraces and retention dams (Morgan, 1995), no-tillage soil management (López & Arrúe, 1997), the use of cover crops (Ingels et al., 1998) and the restriction of agriculture in areas with steep slopes. The use of conservation structures has been widely studied during the past decades (see e.g. Hudson, 1985; Morgan, 1995). More recently, attention has been paid to no-tillage soil management and the application of cover crops.

The effects of no-tillage and minimal tillage on erosion processes, soil quality and crop yields have mostly been investigated for annual cropping systems. Some studies that compared conventional and no-tillage point to equal or higher barley yields with no-tillage (Arshad & Gill, 1997; Cantero-Martínez et al., 2003) and wheat (Agenbag & Maree, 1991; Arshad & Gill, 1997). Just one study was encountered that reports the effect of no-tillage on the yield of a perennial crop. Gómez et al. (1999) observed no significant yield differences during a 15-year trial in an olive orchard in South Spain. The lack of data on perennial crops may be due to the large interannual yield variability of crops like almond and olive and hence the need for long-term records to come to proper conclusions. It is still unclear under which conditions no-tillage leads to an overall decrease in soil erosion. In semi-arid cropping systems, no-tillage requires chemical weeding (i.e. application of herbicides) to prevent weeds to compete for water with the main crop. A drawback of chemical weeding is the risk of pollution of ground and surface water (Lennartz et al., 1997; Cheema & Khaliq, 2000; Landry et al., 2005). Depending on the type and quantity of herbicides applied as well as other local conditions, chemical weeding may be unsustainable. This has for example been the case in Burgundy, France, where drinking water reservoirs have been contaminated with herbicides from vineyards (Landry et al., 2005).

In perennial cropping systems more runoff may be produced under no-tillage than under conventional tillage because of the low soil cover. For example Gómez et al. (2004) measured an increase of soil erosion by water in no-tillage plots in olive orchards. Further investigations are required to find out under which conditions the absence of tillage erosion outweighs the increase in runoff and water erosion. The lowest erosion rates measured by Gómez et al. (2004) were on an orchard floor covered with grass. Little literature is available on the economical differences between systems with no-tillage and conventional tillage in terms of gross margin and profitability. Besides possible yield differences no-tillage may require less operations but includes herbicide costs.

The application of cover crops has a number of advantages relative to no-tillage with herbicides. There is no risk of increased erosion because of a bare soil and no risk of herbicide loss through runoff. In addition, the cover crop will reduce the speed of the runoff and enhance infiltration (Ingels et al., 1998; Pastor, 2004). So far few studies have been conducted on the use of cover crops to mitigate land degradation in semi-arid areas, especially in perennial cropping systems. The remainder of this chapter focuses specifically on that topic.

2.4.3 Vegetation treatments and effects

Possible vegetation treatments to mitigate soil erosion are the application of cover crops and grass- and vegetation strips (e.g. Ingels et al., 1998; Fiener & Auerswald, 2005). Vegetation strips are applied in perennial cropping systems, either in the traffic lane or in the planting line, below the main crop. Cover crops are grown throughout the field and are applied in both annual and perennial cropping systems. More details about the cultivation of cover crops and an overview of cover crop species are provided in §2.2.1 and Appendix 1.

The soil cover provided by vegetation measures protects the soil against splash erosion. The roots increase the resistance of the soil against erosive forces and improve the soil structure (De Baets et al., 2006, 2007). When overland flow occurs, the vegetation stems reduce the flow speed: less soil material is lost and more water infiltrates into the soil (e.g. Ingels et al., 1998; Fiener & Auerswald, 2005). The vegetation litter is decomposed to soil organic matter and humus, which increase the water retention capacity of the soil (Koorevaar et al., 1983). In the case of leguminous cover crops, nitrogen is fixed in the soil (Ingels et al., 1998; Wortmann et al., 2000; Christiansen et al., 2000). Soils covered with vegetation support higher microbiological activity than bare soils (Ingels et al., 1998). Fields that receive organic inputs like cover crop residues have larger and more active microbiological populations (Ingels et al., 1998). The residues of leguminous cover crops are one of the best food substrates for earthworms (Ingels et al., 1998). In vineyards, it has been found that cover crops reduce leafhopper (*Cicadellidae* spp.) densities. However, because of large seasonal variations in leafhopper numbers, cover cropping cannot be relied on for leafhopper control. The cover cropping did not affect total numbers of spiders, which are the most common predators for leafhoppers in vineyards. Furthermore, Ingels et al. (1998) note that few formal studies have been conducted on the effect of cover crops on moth pests, mealy bugs and mites.

An important research question is how the implementation of cover crops affects the production of main crops. Denison et al. (2004), provide figures on the yield of winter wheat in rotation with a fallow year or winter legume cover crop. Based on six year harvest data, it appeared that the wheat production was significantly higher (17%) for the wheat-fallow than for the wheat-legume rotation. This study was located at California, US and involved deep silt loam to silty clay loam soils. Comparable results for barley are provided by a study of Díaz-Ambrona & Mingues (2001). Austin et al. (1998) estimated a yield increase of 15% for barley after a fallow year compared to continuous cropping, as caused by the somewhat higher soil water content at the start of the growing season after fallow. One contradictory finding is that of Christiansen et al. (2000): in Northwest Syria barley yields were somewhat higher in a barley-legume rotation. Overall, the cereal yield differences between rotations with fallow or legume are moderate to small. Díaz-Ambrona & Mingues (2001) point out that cereal-legume rotations might be valuable because they reduce the amount of fertilizer needed and offer an alternative way to control weeds. Finally, it can be stated that literature on the effect of cover crops on the production of olive, almond and grapevine is scarce.

2.4.4 Conditions and thresholds for vegetation

This section aims to illustrate the physical conditions relevant to plant growth in Mediterranean rainfed cropping systems. It focuses on some of the conditions that have to be met when applying vegetation measures against erosion.

Soil properties in the Mediterranean are heterogeneous. Also the annual rainfall depth varies considerably, e.g. from less than 300 mm in south-east Spain to more than 600 mm in the south of France and on the hills of Tuscany, Italy. For this reason, the plants used to mitigate erosion should be selected on a case by case basis. Patterns of soil properties related to topography are important with regard to plant growth, since they determine the spatial distribution of available water. Much research on this topic has been done. Van Wesemael et al. (2000) studied the effect of tillage induced patterns of rock fragment content and soil depth on water balance and tree growth in south-east Spain. It appeared that in cultivated almond plantations soils were deeper and rock fragment content was higher than on slopes under matorral (matorral is a type of shrubland in regions with a Mediterranean climate). A positive relation was found between soil depth and the stem diameter of young almond trees to a maximum of 35 cm soil depth. Soils were deeper in the valleys and showed higher rock fragment cover than on hilltops, resulting in respectively a more stable moisture regime and higher water content (Van Wesemael et al., 2000). Van Wesemael et al. (2003) showed the difference in water redistribution and compared water harvesting strategies in marl and stony slate soils in south-east Spain. On the slates, trees are planted on a regular grid, and the soil management aims at a rapid infiltration of rain into the soil to minimise evaporation. In the traditional system on the marls, trees are planted on the upslope side of retention dams, and rely on the lateral redistribution of runoff and soil moisture (Van Wesemael et al., 2003).

Another aspect of soil moisture is its distribution in time. Western et al. (1998 & 1999) found that soil moisture patterns in the temperate Tarrawarra catchment in south-east Australia show a significant degree of organisation during wet periods as a result of lateral distribution of water. There is little organisation in dry periods. Mellouli et al. (2000) showed the effect of rock fragment content and a stone mulch on evaporation losses in a laboratory study. While the control soil reached half of the total evaporation loss after 4 days, for a soil with higher rock fragment content this was 7 days, and for a soil with stone mulch 8 days.

Soil moisture patterns can also be influenced by vegetation. For instance, Vanderlinden et al. (1998) observed that the infiltration rate was higher under the canopy of olive trees than on the bare soil between the trees. Knowledge on the distribution of available water is crucial to asses the possibility of the application of vegetation strips and cover crops. E.g. it may prove possible to establish vegetation strips at the lower field boundary and not in the middle of orchards: Only at the lower boundary enough water may be available, resulting from the rock fragment mulch (tillage), deeper soil and accumulation of runoff.

In principle, cover crops and vegetation strips compete for water with the main crop. In perennial cropping systems there are two possibilities. The first is that the cover crop does not reduce the available water for the main crop, because it increases infiltration and or uses water from another part of the soil profile than the main crop. The second possibility is that cover crops lead to a decrease in water availability for the main crop and a decrease in production. In that case, dryland farmers will not be inclined to apply such cover crops, unless convincing arguments can be provided that in the long run vegetation measures will benefit production. The only way to get farmers to implement vegetation might be specific and adequate agricultural policy and information. This would involve a compensation for the economic loss associated with the vegetation measures, taking into account the economic value of the cover crops and reduced herbicide and fertilizer needs. For the time being, it remains a question whether cover crops can be incorporated without reducing the water availability to the main crop and if they will benefit production in the long term.

Cover crops and grass strips in olive and almond orchards and vineyards may be present all year round. Cover crops in annual cropping systems can be sown and established during a fallow year, between harvest and the next crop or together with the main crop. Each of these possibilities involves different conditions for the growth of the cover crop. The timing of the presence of the cover crop is rather important in annual cropping systems. For example, it may prove to be possible to establish a cover of weeds in between this years harvest and next years crop in Tuscany, but not in south-east Spain, because of the difference in rainfall and temperature.

2.5 Conclusion

From 1960 to 2001 the surface of perennial crops has increased by 11% in the northern Mediterranean (Spain, Italy, Greece and Portugal), despite a decrease in total agricultural land of about 15%. For the nearby future the trend of a decrease in total agricultural area is expected to continue. The continued mechanisation and intensification during the past decades has given rise to an increase of environmental problems, especially in sloping areas. Practices of intensive tillage, the remodelling of hill slopes and the scarce vegetation cover have promoted increased erosion rates. Agricultural policies play an important role in the occurrence and nature of cropping systems and the associated environmental impacts. Climate change studies point to higher temperatures and higher interannual climatic variability. This may result in a further intensification of land degradation processes.

Possible measures to minimise land degradation in dryland agriculture are the practice of reduced or no-tillage, the construction of conservation structures as terraces and retention dams and the use of vegetation. The amount of literature on no-tillage practices and vegetation measures is limited, especially for perennial crops. At present the application of cover crops and no-tillage is rare in the Mediterranean. Some more information and examples are available from California, USA. Nevertheless, it remains unclear under which conditions vegetation measures can and cannot be applied successfully in rainfed cropping systems, both in physical and economical terms. Little is known about the water requirements of cover crops and their effects on the productivity of the main crop. Because of the large interannual variability of climate and yields long term experiments are required to study these issues. The use of water harvesting techniques may increase the water availability and enable vegetation measures under drier conditions.

CHAPTER 3
WATER AVAILABILITY IN ALMOND ORCHARDS ON MARL SOILS: THE ROLE OF SOIL EVAPORATION AND RUNOFF

This chapter has been accepted for publication in Journal of Arid Environments: Meerkerk, A.L., van Wesemael, B., Cammeraat, E. Water availability in almond orchards on marl soils in south east Spain: The role of evaporation and runoff.

3.1 Introduction

Rainfed orchards cover a significant part of the land in the Mediterranean region. For example olive (*Olea europea* L.) plantations occupy 5-80% of the agricultural land (excluding pastures) in the south of Spain, Italy and Greece, and 10-55% in Tunisia (Fleskens, 2007). A large part of the plantations is located on marginal, sloping lands. The area planted with olive and almond (*Prunus dulcis* Mill.) has increased continuously over the past decades. The largest increase in olive plantations during the period 1990-2000 occurred in Spain (+6%), Morocco (+27%) and Syria (+21%, Fleskens 2007). In this chapter we focus on Murcia province in south-east Spain, which forms a representative area of marginal rainfed orchards. A large part (86%) of the almond and olive plantations in Murcia are rainfed (CREM, 2006), where the annual rainfall is in the order of 300 mm. The cropping system has changed considerably over the past decades, following the mechanisation and intensification of agriculture (Beaufoy, 2003; Faulkner et al., 2003; Fleskens, 2007). The intensification has lead to accelerated soil erosion due to tillage (Poesen et al., 1997; van Wesemael et al., 2006), removal of soil and water conservation structures (Faulkner, 1995) and land levelling (Poesen and Hooke, 1997; Borselli et al., 2002).

Traditionally, terraces and earthen dams allowed runoff water to infiltrate and to be exploited by the crops. Instead of relying on runoff, trees are now planted on a wide grid, maximising the root volume and the uptake of soil moisture (van Wesemael et al., 2003). Clean sweeping (i.e. frequent shallow tillage) is applied to eliminate transpiration losses by weeds and to promote infiltration, in order to increase the water availability for the trees and the orchard productivity (Poesen et al., 1997; Pastor, 2004). The number of tillage passes per year ranges from one to two in low-input, extensive plantations to five in intensive systems (Poesen et al., 1997; Beaufoy, 2003). The growth of weeds and the number of tillage passes required increases with rainfall depth and frequency (Beaufoy, 2003). The depth of tillage depends on the specific equipment used and is typically in the order of 15 to 25 cm (Pastor, 2004).

Despite the widespread application of clean sweeping, there is a lack of quantitative data on its effects on soil evaporation and the water availability for the trees. A potential drawback in dry areas is the fact that tillage can lead to an increase of evaporation and a decrease in water availability (Pastor, 2004). Evaporation and runoff form an undesirable loss of water from a producer's perspective because this water cannot be used by the trees. Evaporation and runoff therefore limit the production of biomass. Crop transpiration, the only term of the water balance that is directly related to harvested biomass, can be as low as 5-25% of rainfall and evaporation as high as 50% (Rockström and Valentin, 1997; Rockström et al., 1998;

Wallace, 2000). The annual runoff coefficient in frequently ploughed olive groves in the Mediterranean is in the order of 0-18% (Kosmas et al., 1997; Gómez et al., 2004). Field data presented by Pastor (2004) show that the water use efficiency and yields can be increased by switching from clean sweeping to no-tillage. Other studies show a neutral to positive effect of no-tillage treatments on yield and soil moisture (Gómez et al., 1999; Hernández et al., 2005). The 15-year study by Gómez et al. (1999) showed no overall yield difference between a no-tillage and conventional tillage treatment. Nevertheless, during a dry year with 406 mm of rainfall, the yield of the no-tillage treatment was about twice that of the control treatment. The authors suggested that the difference might be due to the absence of roots in the surface layer of the conventional treatment. Hence it seems that clean sweeping may not be the most effective strategy to increase the water availability under dry conditions (annual rainfall < 400 mm y^{-1}).

We hypothesise that the practice of clean sweeping leads to large evaporation losses in rainfed orchards receiving less than 400 mm of annual rainfall. The objective of this paper is to determine the effect of evaporation from the plough layer and overland flow on the water availability in almond orchards on loamy soils in south-east Spain. It deals with the impact of evaporation and runoff on water availability, using a calibrated water balance model, field observations, and a spatially explicit estimation of runoff and run-on. The study is done in duplicate to ensure that the observations are representative. Given the high rainfall variability in the Mediterranean region (Mulligan, 1998), we address both dry and wet years occurring within the economic life time of an orchard. Finally, the implications for managing these vulnerable cropping systems are discussed.

3.2 Materials and methods

3.2.1 Study site

The orchards are located in the Cárcavo basin in Murcia, Spain (38.20° N, 1.52° W). Some of their properties are listed in Table 3.1.

Table 3.1. Characteristics of the orchards.

	C1	C2
Area (ha)	1.51	2.06
Tree spacing (m)	7.5 x 7.6	6.2 x 6.2
Elevation (m)	405	328
Mean slope (°)	3.57	4.56
Longitude (°)	-1.51204	-1.52266
Latitude (°)	38.17270	38.20322

The area receives an annual rainfall of 298 mm at a mean annual temperature of 16.4° C (Almadenes station, period 1966-2000, 38.23° N, 1.55° W). The climate is characterised by a high interannual rainfall variability and a reference evapotranspiration of around 1125 mm y^{-1}, as calculated with the Hargreaves equation (Hargreaves and Samani, 1985). The geology of the area consists of Cretaceous marls and limestone with some sandstone outcrops. The orchards are on marl substrate and their soils are classified as Calcaric Regosols with an organic matter content of approximately 6 g kg^{-1} (ICONA, 1986; Table 3.2). More details on the Cárcavo basin are provided by Castillo et al. (2007) and Lesschen et al. (2007).

Table 3.2. Average soil properties of the upper 40 cm of the soil in orchards C1 and C2. The albedo figures refer to the soil surface.

	Fine earth [a]							Albedo	
	Sand [b] ($g\ kg^{-1}$)	Silt [b] ($g\ kg^{-1}$)	Clay [b] ($g\ kg^{-1}$)	Bulk density ($Mg\ m^{-3}$)	$CaCO_3$ ($g\ kg^{-1}$)	Stones ($g\ kg^{-1}$)	Porosity ($m^3\ m^{-3}$)	Dry	Wet
C1	363	504	133	1.3	550	0	0.50	0.19	0.11
C2	336	531	133	1.2	656	19	0.51	0.26	0.17

[a] The fine earth is the fraction of soil particles with a diameter <= 2 mm.
[b] Grain size fractions per kg of fine earth: clay < 2 µm; silt 2-63 µm; sand 63-2000 µm.

The plantation of C1 was about five years old in 2005 and the plantation of C2 three years. Several soil and water conservation structures are present in the orchards, which is a common feature in the area. C1 has one terrace in the north west corner, another terrace in the north east and a ditch (response unit C1d) that collects runoff from the small hills in the north and drains to the lower terrace (Fig. 3.1, next page). C2 has a terrace along the field diagonal.

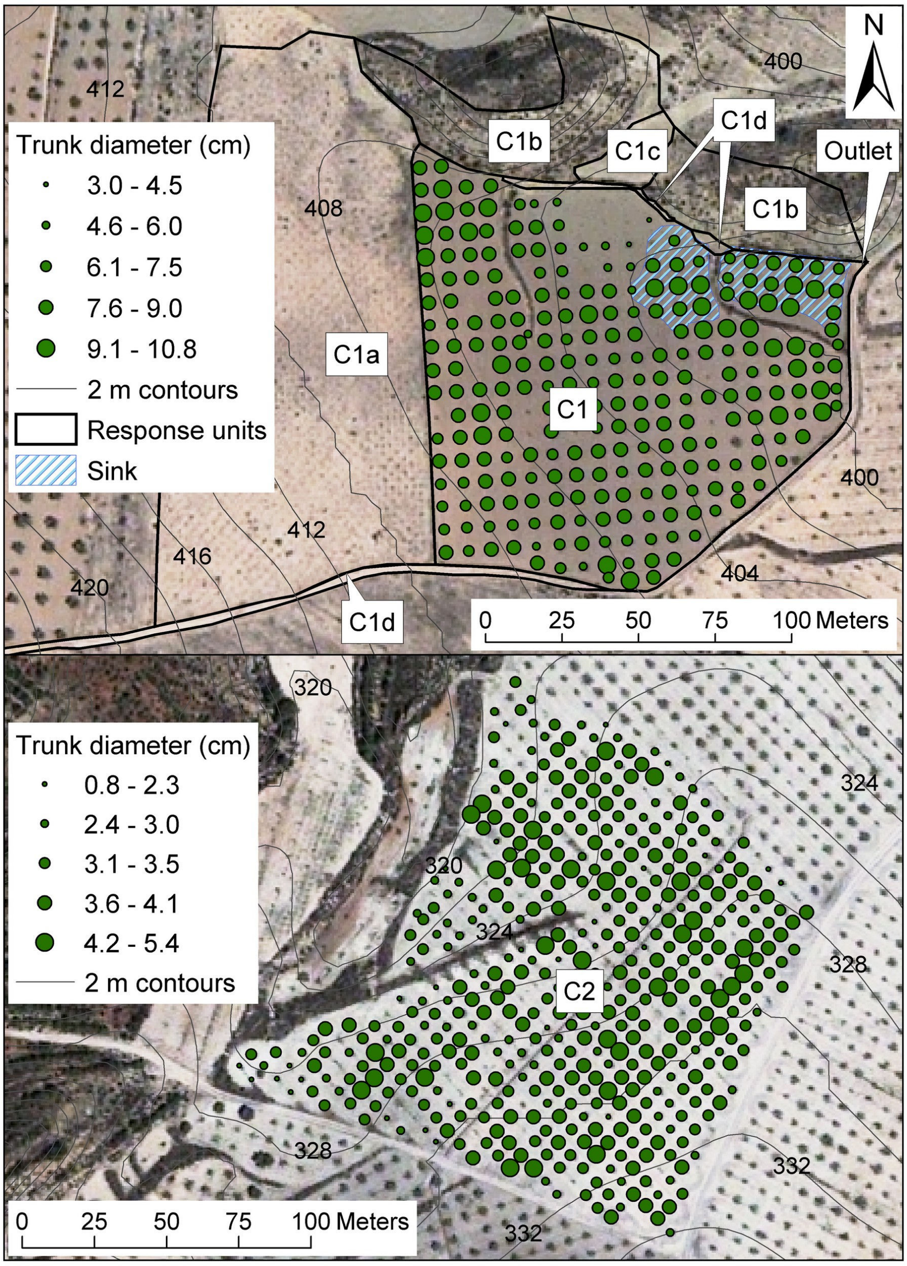

Figure 3.1. Overview of orchard C1 (38.1727° N, 1.51204° W) and C2 (38.20322° N, 1.52266° W). The hydrological response units are described in Table 3.3 (§3.2.6).

3.2.2 Measurement of orchard properties

The trunk diameters were measured in 2004, in order to detect spatial patterns in tree growth related to overland flow. The diameter was measured at a height of 30 cm using sliding callipers. The tree canopy cover was measured in orchard C1 in September in order to produce an estimate of the crop transpiration. Photos of the canopy were captured with a camera mounted beneath a helium balloon. The image pixel size obtained was 2 cm^2 or less. Seven images were selected randomly throughout the orchard and subjected to a supervised maximum likelihood classification of canopy cover. The corresponding classification error of canopy cover was approx. 1%. A total of 42 root observations were done in September, in order to determine the presence of tree roots in the top soil (orchard C1). The observations were made in soil pits at distances of 0.5, 1, 2 and 4 m from randomly chosen trunks. The number of roots was counted visually for vertical 10 by 10 cm squares of soil at a depth of 5-15 and 15-25 cm.

3.2.3 Modelling of the water balance

In order to quantify the water balance of the plough layer, we assumed that it contains no roots as a consequence of clean sweeping. The evaporation from a 15 cm plough layer (E_p) can now be expressed as (Eq. 3.1):

$$E_p = P + Q_{on} - D_{15} - \Delta S - Q_{off} \qquad (3.1)$$

Where P is precipitation, Q_{on} is run-on, D_{15} is drainage below 15 cm, ΔS is the change in soil moisture storage for a given time interval and Q_{off} is runoff. All terms expressed in mm. A schematic representation of the water balance is given in Figure 3.2.

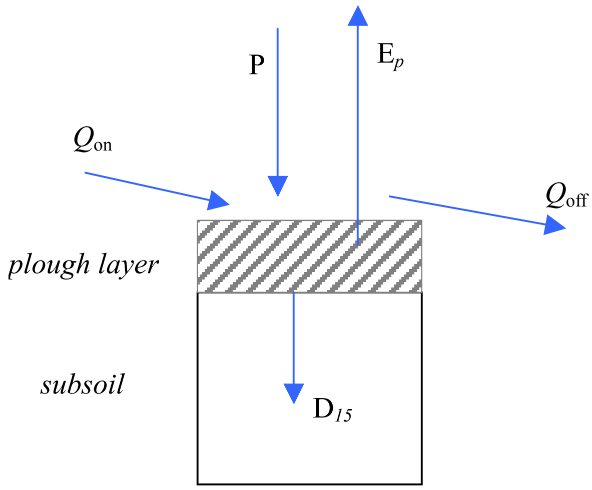

Figure 3.2. Schematic representation of the water balance of the plough layer (Eq. 3.1).

In order to obtain E_p, the soil water balance was simulated using the BUDGET model (Raes et al., 2006). The soil was represented in the model by 7 horizontal layers of 5 cm thickness. For each layer BUDGET calculates the vertical flux of soil water at the upper and lower boundary as well as the net change in moisture storage. The model calculates the components of the water balance at a daily time step for each layer. Drainage is calculated first, based on the final soil moisture content of the preceding day, followed by infiltration and finally evaporation. The main inputs of the model are rainfall, adjusted for runoff losses or gains, reference evapotranspiration (ET_0, mm) as calculated by the FAO Penman-Monteith equation (Allen et al., 1998), soil porosity, field capacity, wilting point (i.e. the soil moisture content at a matric potential of -1.5 MPa), τ_d [-], which determines the rate of drainage (Raes, 2002), and hydraulic conductivity (K_{sat}, mm h^{-1}). Please note that the τ of Raes (2002) is renamed to τ_d, in order to distinguish it from $\tau(h)$ in Chapter 5. The model output consists of the daily water fluxes of the 5 cm layers and of the entire profile, as well as soil moisture storage. An additional model parameter is the depth to which the soil contributes to evaporation.

It can be noted that a one dimensional (1D) modelling approach was chosen (Eq. 3.1) because this offers a number of advantages compared to more complex 2D and 3D models. The relative simplicity of a 1D model is practical in terms of model development, calibration and validation. A 2D or 3D approach requires more input data to represent the lateral variability in soil and surface properties. Furthermore, additional algorithms are required to describe the redistribution of runoff and soil water. Another advantage of a 1D approach is that the water balance of different locations can be easily compared.

3.2.4 Lysimeter experiments

BUDGET was calibrated by a lysimeter experiment carried out in the laboratory, which lasted seven months. The experimental set-up is shown in Figure 3.3.

Figure 3.3. Experimental set-up of the lysimeter experiment.

Soil monoliths from orchards C1 and C2 with a diameter of 30 cm and a height of 35 cm were installed on electronic balances connected to a data logger. Simulated rain events of 10 to 115 mm were applied at an average intensity of 40 mm h^{-1}, using deionised water. Wind was simulated by a fan at 0.4 m s^{-1} for column C1 and 1.5 m s^{-1} at C2. Other parameters monitored were air temperature, relative humidity, incoming radiation and drainage. The latter was measured by placing the drainage tubes of the soil columns in separate tipping bucket rain gauges, which were connected to the data logger. The soil columns were illuminated 12 hours a day at around 360 W m^{-2}, which is comparable to the average incoming radiation in Murcia. Following the lysimeter experiment, BUDGET was calibrated for the entire 35 cm profile. Three parameters were adjusted for each column: the soil depth that contributes to evaporation, moisture content at wilting point and τ_d. The other soil parameters were determined from laboratory measurements (porosity, field capacity, wilting point) and rainfall simulations (K_{sat}; §3.2.5; Table 3.3).

The contribution of the subsoil (depth > 25 cm) to evaporation was studied with another lysimeter experiment. Two air-dry columns with a diameter and depth of 30 cm were saturated with water at a depth of 25-30 cm. This was done by placing them in a water tank until they had soaked up the amount of water required to saturate this layer of known porosity. Subsequently, the columns were illuminated 12 hours per day at 376 W m^{-2} and a fan provided a wind of 1.4 m s^{-1}. The same variables were monitored as described for the calibration experiment.

3.2.5 Runoff measurements

The runoff response of the orchards was determined by means of rainfall simulations, which were carried out in January and September. A drip-plate (Amsterdam type) simulator was used for plots of 50 x 100 cm as described by Bowyer-Bower and Burt (1989). Demineralised water was applied for a minimum of 30 minutes at an intensity of 10 to 70 mm h^{-1} (average of 42 mm h^{-1}). Use was made of demineralised water was because infiltration rates are significantly overestimated when using solute-rich water (tap water; Borselli et al., 2001). It was attempted to apply 20, 40 and 60 mm h^{-1} in three adjacent plots, however it was difficult to regulate the intensity accurately. I_a was measured as the rainfall applied until the start of runoff production. K_{sat} was estimated by the asymptote of the infiltration curve, which shows the decline in infiltration rate in time. A total of 18 simulations was done, nine in each orchard, which included two distinct surface conditions: the very loose surface structure common after tillage and the surface with a structural crust, which is formed during rainfall (Valentin and Bresson, 1997). Furthermore, three simulations were executed in an abandoned vineyard upslope of orchard C1 (Fig. 3.1, response unit C1a). Soil moisture was measured before and during rainfall simulations using a theta probe (Delta-T Devices Ltd.), which was installed horizontally at a depth of 5 cm at the centre of the plot edge (i.e. the 100 cm edge, aligned in slope direction).

Traces of concentrated flow and reinfiltration were mapped after natural rain storms in 2005 and 2006, in order to determine the influence of overland flow on the spatial distribution of available water. The traces were mapped with a GPS device. Rainfall data with a resolution of 1 minute was available from a tipping bucket rain gauge located at 1.4 km from orchard C1.

3.2.6 Runoff modelling

A distinction was made between surfaces with different infiltration characteristics, so called hydrological response units (HRUs), which is a common approach in runoff modelling (Beven, 2001). Runoff production was estimated for each HRU separately. The HRUs were identified on the basis of vegetation cover, stone cover, crusting stage and infiltration characteristics (Fig. 3.1 & Table 3.3). Vegetation and stone cover were derived from the aerial photographs (Fig. 3.4).

Table 3.3. Hydrological properties and soil cover of the hydrological response units (HRUs). C1, C2: almond fields, C1a: abandoned vineyard, C1b: rock outcrop, C1c: unused patch, C1d: dirt track / gully.

HRU	Soil crust typology [a]	K_{sat} (mm h^{-1})	I_a [e] (mm)	θ_i [b] (m^3 m^{-3})	n	I_{10} threshold [c, f] (mm h^{-1})	Vegetation & stone cover (%) Range	Vegetation & stone cover (%) Ave.
C1	Ploughed	29.6 [b]	9.7 [b]	0.08	3	< 19	1-5	3
	Structural	28.8 [b]	7.3 [b]	0.11	6			
C1a	Depositional	19.3 [b]	3.8 [b]	0.15	3	< 19	0-100	10
C1b	Depositional	20 [d]	25.2 [c]			< 19	10-100	60
C1c	Structural - depositional	24 [d]	25.9 [c]			19	1-5	2
C1d	Depositional	19 [d]	3.8 [d]			< 19	1-5	2
C2	Ploughed	24.4 [b]	7.2 [b]	0.05	6	< 19	1-4	3
	Structural	18.3 [b]	1.0 [b]	0.10	3			

[a] According to Valentin and Bresson (1992); [b] Rainfall simulation; [c] Field observation of five storms; [d] Estimate; [e] Initial abstraction (mm); [f] threshold of the ten minute peak intensity of a storm for runoff production.

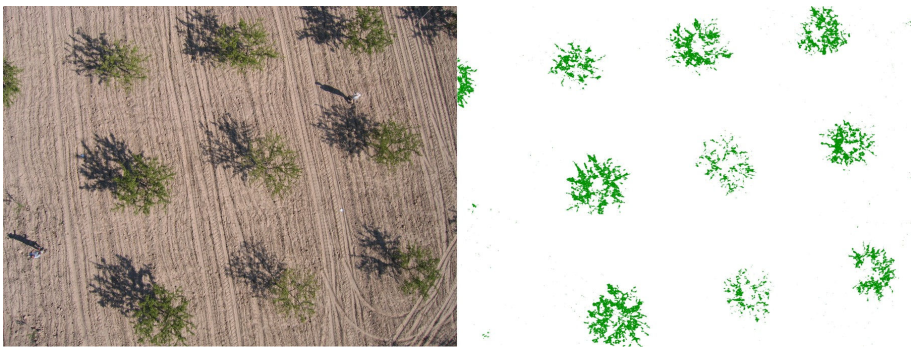

Figure 3.4. Example of the aerial imagery captured in orchard C1 (left) and the corresponding canopy cover classification (right).

The spatial redistribution of runoff was addressed only for orchard C1, because there was little evidence of concentrated flow and reinfiltration in C2. The HRUs at C1 were further subdivided into flow units based on topography to enable the routing of overland flow. Use was made of a digital elevation model with a cell size of 5 m, which was derived from the 1:5000 topographic map. The extent of the catchment consisting of C1 and connected upslope areas was determined from the observed pathways of overland flow (Fig. 3.5).

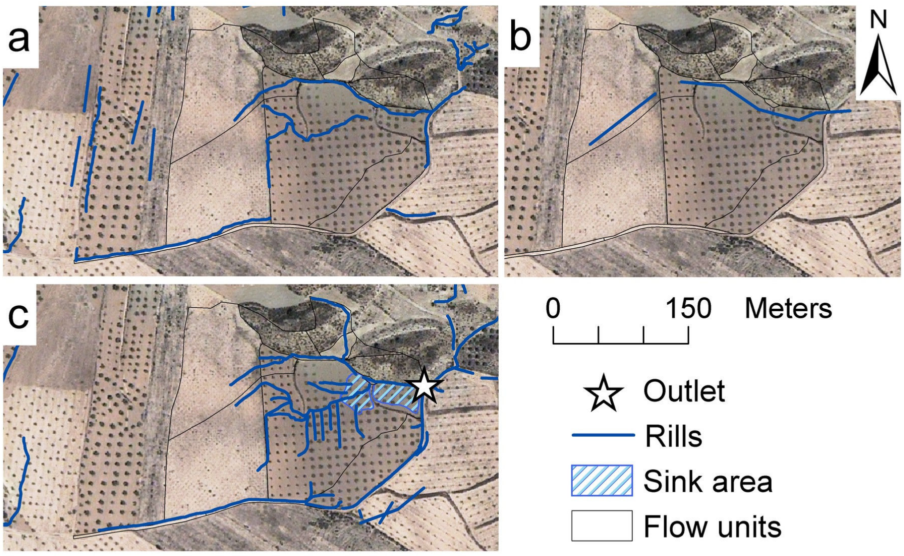

Figure 3.5. Rills observed at orchard C1 for three rain events. (a) P = 23.2 mm; I_{10} = 67.5 mm h^{-1}. (b) P = 25.9 mm; I_{10} = 19.2 mm h^{-1}. (c) P = 92.8 mm, I_{10} = 55.9 mm h^{-1}.

At first, a simple runoff estimation was done using the curve number (CN) method (Soil Conservation Service, 1985). A drawback of the CN method is that it calculates runoff on an event basis and ignores the impact of rainfall intensity and its variability during the event (Mockus, 1964 as cited in Mishra and Sing, 2003, p. 129). This leads to unreliable results in semi-arid areas, where the within storm variability of rainfall intensity has a large influence on runoff production (Nicolau et al., 1996; Castillo et al., 2003; Reaney et al., 2007). Observations in orchard C1 point out that these findings apply to the Cárcavo basin as well: the events of Figure 3.5a and b have a similar rainfall depth (23-26 mm), but the ten minute peak intensity

(I_{10}, mm h^{-1}) of the event of Figure 3.5a is about three times larger than that of Figure 3.5b and results in considerably more rills and runoff. The CN method however produces a larger amount of runoff for the event of Figure 3.5b. It was therefore decided to use an alternative approach which allows to incorporate the effect of rainfall intensity. The approach is based on an hourly time step because of the hourly rain data available for modelling. The input consists of hourly rainfall (P, mm), the initial abstraction prior to runoff (I_a, mm), and the saturated hydraulic conductivity (K_{sat}, mm h^{-1}), which forms an alternative for the potential maximum retention (S, mm) of the CN method (Mishra and Sing, 2003). I_a is the sum of surface storage and initial infiltration. P is split in two parts, distinguishing the rain that falls at peak intensity (I_{10}) and the amount that falls during the remaining 50 minutes (P_{50}, mm). The hourly runoff (Q_{off}, mm) is now estimated by (Eq. 3.2):

$$Q_{off} = \frac{1}{6}\left(I_{10} - K_{sat}\right) + \bar{t}_{50}\left(\frac{P_{50}}{\bar{t}_{50}} - K_{sat}\right) \text{ for } P > I_a \qquad (3.2)$$

The first term of Eq. 3.2 returns the runoff produced during the 10 minute rainfall peak. The constant $\frac{1}{6}$ h serves to obtain Q_{off} in mm. The second term represents the runoff produced during the off-peak period, where $\bar{t}_{50}$ is the average time of rainfall during this period (h). The effect of initial soil moisture (θ_i, m^3 m^{-3}) and soil sorptivity on infiltration is ignored. It is assumed that θ_i is low and uniform and that I_a is constant. The first term of Eq. 3.2 is set to zero if $I_{10} \leq K_{sat}$ and the second term if $\frac{P_{50}}{\bar{t}_{50}} \leq K_{sat}$. Values of I_{10} are estimated by a regression of I_{10} and P (expressed in mm h^{-1}) using a data set from Las Caras station in Cárcavo (Fig. 3.6).

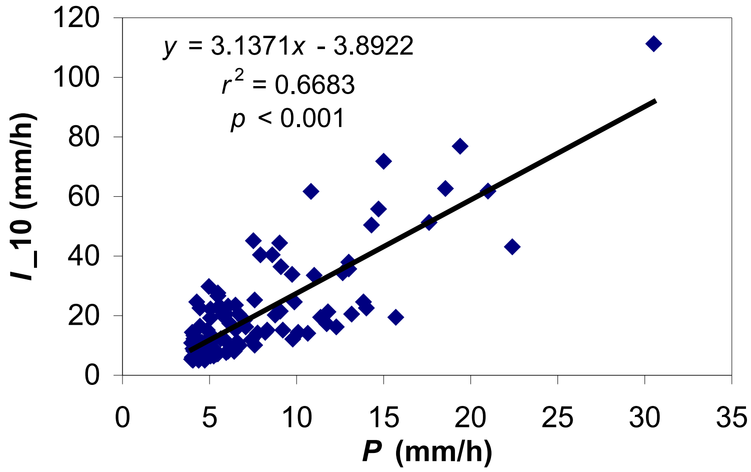

Figure 3.6. Regression of ten minute peak intensity (I_{10}) and hourly rainfall (P). The data set consists of eight years of rainfall measurements at Las Caras station.

This set covers eight years of rainfall at ten minute resolution and shows a reasonable linear fit ($r^2 = 0.66$, $p < 0.001$; Eq. 3.3):

$$I_{10} = 3.137\,P - 3.892 \qquad (3.3)$$

During most hours with rainfall it does not rain every minute of the off-peak period due to the small size and erratic distribution of the rainfall. Accordingly, episodes with a larger than average rainfall intensity are common. The use of the average off-peak intensity would therefore result in an underestimation of Q_{off}. Variable $\bar{t}_{50}$ (Eq. 3.2) is an attempt to reduce this underestimation. We know that $t_{50} \in \left[\frac{P_{50}}{I_{10}}, \frac{5}{6}\right]$ while the intensity during the off-peak period cannot exceed I_{10}. Under the assumption that t_{50} has a symmetrical distribution we can calculate $\bar{t}_{50}$ using (Eq. 3.4):

$$\bar{t}_{50} = 0.5\left(\frac{5}{6} + \frac{P_{50}}{I_{10}}\right) \qquad (3.4)$$

Eq. 3.4 returns values of $\bar{t}_{50}$ that range between 25 and 50 minutes. For the majority of the events, $\frac{P_{50}}{\bar{t}_{50}}$ is now larger than the average rainfall intensity of the off-peak period (Eq. 3.2).

3.2.7 Model application

The water balance of the plough layer was simulated using the calibrated water balance model and the runoff estimation method described previously (Eq. 3.2). A long term series of hourly weather data were used from Barranco de la Alquería, which is located 52 km south-west of Cárcavo (37.78° N, 1.83° W; Cammeraat, 2004). The years 1997 ("wet"), 1999 ("average") and 2005 ("dry") were selected for modelling, having an annual rainfall of respectively 429, 287 and 169 mm. The wet and dry year represent the range of annual rainfall in Cárcavo with a recurrence interval of 7.7 years (Almadenes station, 1966-2000). BUDGET was run for the 35 cm profiles of orchards C1 and C2 for the wet, average and dry year. E_p and D_{15} were derived from the simulated water fluxes of the 5 cm layers. Simulations were executed for different topographic positions in the orchards: a reference location where runoff has no effect ($Q_{on} = Q_{off}$) and a slope position which produces Q_{off} and does not receive Q_{on}. Furthermore, a sink area was defined where $Q_{on} > Q_{off}$, based on observed ponding of runoff and sediment deposition in C1 (Fig. 3.5c). The sink area consists of a depression where significant amounts of run-on infiltrate. The flow towards the sink was estimated by routing the Q_{off} from upslope flow units at an hourly time step. It was assumed that Q_{off} from activated flow units infiltrates on down slope units if those do not produce Q_{off} themselves, i.e.

where $P < I_a$ or $I_{10} < K_{sat}$. During some storms, the modelled inflow exceeded the capacity of the sink. The water was then drained by the outlet, as observed in the field (Fig. 3.5c). The capacity of the sink was set at 60 mm in accordance with the height of the rim along the lower field border.

The K_{sat} and I_a input for runoff estimation (Eq. 3.2) of HRUs C1, C1a and C2 was derived from the rainfall simulations (Table 3.3). In the case of C1 and C2, the average value of the ploughed and crusted surface was used. The runoff response of C1d was assumed to be the same as that of C1a. For the remaining units (C1b & C1c), I_a values were estimated by the rainfall threshold for runoff production as observed for five storms during 2005-2006 (Fig. 3.5, Table 3.3). The K_{sat} used for C1b was an educated guess. The vegetation cover of unit C1b consists mostly of *Stipa tenacissima* (L.) and is relatively high (20%). It was assumed that only 80% of the C1b surface contributes to runoff due to infiltration at the *Stipa* tussocks. The K_{sat} of C1c, a patch of bare land without regular tillage, was estimated as the average of the abandoned vineyard (C1a) and orchard C1.

3.3 Results and discussion

3.3.1 Orchard properties

The classification of the aerial photos shows that the canopy cover in orchard C1 ranges from 0.9 to 5.2%, with an average of 3.2% (Fig. 3.4). During winter the trees are leafless and the cover is close to zero. The root survey showed that the 5-15 cm layer contains no roots in 33% of the pits. 64% have 1 to 10 fine roots and 2.4% have more than 10 roots. Furthermore, 90% of the pits have no roots with a diameter of one mm or more in this layer. There are no pits with more than two roots of that size. Sometimes the roots could be related to weed remnants, supporting the conclusion that the plough layer (0-15 cm) contains few tree roots. Most of the water stored in the plough layer is therefore lost by evaporation from the bare soil. Transpiration by weeds is of minor importance because their growth is sparse and limited to the short period between rainfall and the next tillage pass.

At first sight, the spatial variability in tree trunk diameter appears to be random (Fig. 3.1). A closer look suggests that there is a concentration of larger trunks in the northeast and northwest of orchard C1. Both locations have a topography with a relatively concave curvature and a downslope barrier for overland flow, consisting of elevated terrace borders. In the northeast the large trunks overlap with the observed sink area of overland flow. The implications in terms of run-on and water availability are discussed in §3.3.4.

3.3.2 Lysimeter results and model calibration

The lysimeter experiment shows higher evaporation rates for profile C2 than for C1 during the first four rainfall-evaporation cycles (Fig. 3.7). The reason is that the rain on both columns was continued until the start of drainage and the corresponding rainfall quantity was larger for C2.

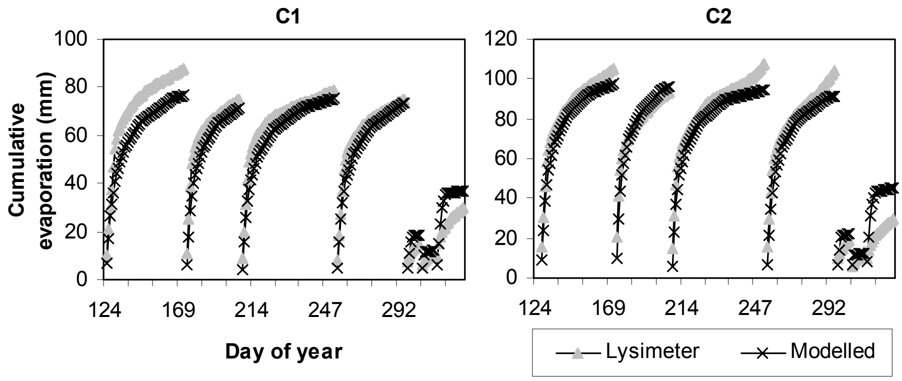

Figure 3.7. Calibration of the BUDGET model for evaporation as measured with lysimeters for two silt loam soils.

The separate experiment with a saturated layer at 25 cm depth showed that the evaporation from the subsoil was 3.9 mm (18% of the water applied) after 64 days for profile C1 and 5.2 mm (30% of the water applied) after 53 days for profile C2. These results indicate that the evaporation from depths greater than 25 cm is rather limited. It can be argued that the evaporation from the subsoil will be even less under field conditions, due to water uptake by the tree roots.

The calibration of the BUDGET model was satisfactory (Fig. 3.7, Table 3.4). The root mean square error in soil moisture was smaller than 0.046 m^3 m^{-3}. The model overestimated the rate of evaporation during the energy limited stage after rainfalls in the order of 20 to 40 mm (Fig. 3.7, from *DOY* 292 onwards; see Ritchie (1972) for an explanation of the energy limited stage of evaporation, also referred to as first stage). This error is not very important when considering the cumulative evaporation over longer periods (Table 3.4).

Table 3.4. Calibration of the water balance model for the C1 and C2 profiles. RMSE stands for root mean square error.

		I [a]	D [b]	E_s [c]	ΔS [d]	Soil moisture (m^3 m^{-3})	
		(mm)	(mm)	(mm)	(mm)	RMSE	Max error
C1	Measured	415	23	372	20		
	Modelled	415	26	370	19	0.045	0.075
C2	Measured	504	20	475	9		
	Modelled	504	19	475	10	0.031	0.075

[a] Infiltration; [b] Drainage; [c] Soil evaporation; [c] Change in soil moisture storage

For both soil profiles, the best calibration was obtained by fixing the soil depth that contributes to evaporation at 30 cm and $\tau_d = 1$. For lower values of τ_d the simulated drainage was too small compared to the observations. The soil moisture content at wilting point was adjusted from 0.13 to 0.15 m^3 m^{-3} for C1 and from 0.08 to 0.07 m^3 m^{-3} for C2, which improved modelled evaporation. The wilting point values used initially (0.13 and 0.08) were a rough estimate based on the range of soil moisture observed during the lysimeter experiment.

3.3.3 Model runs with observed weather data

The simulated E_p in the wet, average and dry year ranges from 70 to 104% of annual rainfall, depending on topographic position (Table 3.5). Note that evaporation can exceed rainfall when runoff water is concentrated in a small area.

Table 3.5. Water balance of the 15 cm plough layer and 35 cm profiles of orchard C1 and C2 at different topographic positions. All figures in mm.

					Plough layer			Profile		
	Year	Q_{on}	I	Q_{off}	D_{15}	E_p	ΔS	D	E_s	ΔS
C1-reference	Wet	-	429	-	130	299	0	67	361	1
	Average	-	287	-	41	245	1	0	287	0
	Dry	-	169	-	0	169	0	0	169	0
C1-slope	Wet	-	416	13	117	298	1	54	362	0
	Average	-	285	2	39	245	1	0	285	0
	Dry	-	166	3	0	166	0	0	166	0
C1-sink	Wet	372	611	189	286	325	0	223	389	-1
	Average	61	348	0	102	245	1	41	306	1
	Dry	92	250	11	73	177	0	36	217	-3
C2-reference	Wet	-	429	-	124	304	1	63	366	0
	Average	-	287	-	42	245	0	0	286	1
	Dry	-	169	-	0	169	0	0	169	0
C2-slope	Wet	-	407	22	105	301	1	45	362	0
	Average	-	282	4	40	241	1	0	284	-2
	Dry	-	163	6	0	163	0	0	164	-1

When considering the entire 35 cm profile evaporation equals 84 to 128% of annual rainfall. Values of D_{15} range from 0 to 66% of annual rainfall. The large evaporation and small D_{15} result from the limited rainfall depth per event. The largest daily rainfall is 45 mm and occurs during the wet year (Fig. 3.8).

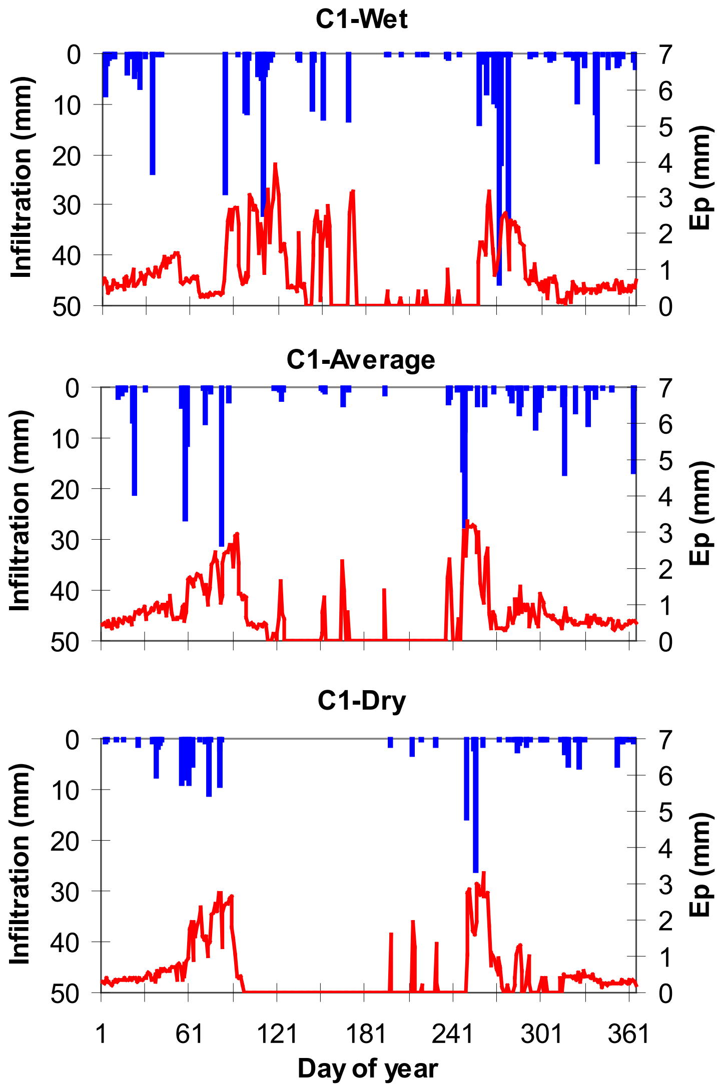

Figure 3.8. Daily infiltration and evaporation (E_p) for a 15 cm plough layer during a wet, average and dry year, as simulated for the topographic reference position of orchard C1.

During the dry year, the largest rain event is 26 mm and no D_{15} is produced on the reference and slope position (Table 3.5). A second explanation for the limited drainage in dry years is the lower soil moisture content at the start of rainfall. This is illustrated by the average moisture content of the upper 5 cm of the soil, which is 17% in the wet year, 16% in the average year and 14% in the dry year (run C1-reference, results not shown). The maximum evaporation rate from the plough layer is higher during the wet year as well, reaching 4 mm day^{-1} (Fig. 3.8). The simulation results are rather similar for the soils from orchards C1 and C2. The only clear difference is the lower infiltration capacity and higher runoff at C2 (Table 3.5).

The results indicate that the evaporation from the plough layer is considerable. They confirm the observation in previous publications that percolation to deeper soil layers minimises evaporation losses (van Wesemael et al., 1996; Pastor, 2004; Tubeileh et al., 2004).

3.3.4 The role of runoff

Five runoff events were recorded in field C1 and three events in C2 during 2005 and 2006. Figures 3.5a and 3.5b illustrate the importance of rainfall intensity rather than quantity for runoff generation during light rainfall. The infiltration characteristics and runoff thresholds of the HRU are listed in (Table 3.3). The infiltration capacity of the almond fields decreases after crust formation. A single rainfall is sufficient to produce a new crust after tillage. The I_a of the crusted soil at C2 is remarkably low at 1 mm and was observed for a rainfall intensity between 12 and 57 mm h^{-1}. The difference between C1 and C2 may be related to the higher silt and calcium carbonate content in the soil of C2 (Table 3.2). The rock outcrop (C1b) and the unused patch of land (C1c) have relatively high I_a values, which seems related to the high vegetation and stone cover at C1b and the rather faint slope of C1c (Table 3.3). From the rainfall simulations it appears that I_a is not related to θ_i for $\theta_i \in [0.03, 0.18]$ (r^2 = 0.02, n = 25) and that θ_i is generally small (Table 3.3). For 44% of the runoff events of the BUDGET runs, the θ_i for position C1-reference is below 0.18 m^3 m^{-3} and the average is 0.19 m^3 m^{-3} (results not shown). It therefore appears that the omission of θ_i in the runoff calculation (Eq. 3.2) does not influence the results.

The simulated amounts of runoff are small and the annual runoff coefficients of the HRUs range from 0.7 to 6.3 % (Table 3.6). The variability in runoff response illustrates the importance of rainfall intensity. The results are in line with other publications on semi-arid catchments, which have related the erratic runoff response to surface properties and storm characteristics (Nicolau et al., 1996; Puigdefabregas et al., 1998; Castillo et al., 2003; Reaney et al., 2007). The effect of spatial patterns is most pronounced when the rainfall intensity is in the same order of magnitude as K_{sat}, so that the spatial distribution of surface properties results in a dynamic pattern of source and sink areas (Western et al., 2001; Castillo et al., 2003).

Table 3.6. Annual runoff production of the hydrological response units (Fig. 3.1) as simulated with the weather data from Alquería station (Eq. 3.2). Annual rainfall is 429 mm for the wet year, 287 mm for the average year and 169 mm for the dry year. The number of runoff producing rain events is added between brackets.

	Runoff year sum (mm)				
Year	C1	C1a; C1d	C1b	C1c	C2
Wet	13.3 (3)	26.9 (9)	16.1 (3)	14.7 (2)	22.4 (7)
Average	2.0 (1)	6.0 (4)	2.9 (2)	2.8 (2)	4.8 (3)
Dry	3.2 (1)	7.4 (3)	3.0 (1)	4.1 (1)	6.1 (3)

The runoff coefficient of the entire micro-catchment of C1 is quite low, being 0.27% for the dry year, 0% for the average year and 1.8% for the wet year. The reason is that a large part of the overland flow is concentrated in the sink area and infiltrates there (Fig. 3.5c). Hence the net run-on contribution in the sink area equals 21 to 47% of annual rainfall (Table 3.5). The trees respond to the increased water availability, since the trunk basal area is 12% larger ($p < 0.03$) compared to the rest of the orchard. Similar observations were made for rainfed olive orchards in Córdoba by Castro et al. (2006), who argue that the redistribution of rainfall may have important implications for the water balance and orchard productivity.

The use of K_{sat} as the fixed infiltration capacity of the soil implies a slight overestimation of Q_{off} in Eq. 3.2: the infiltration capacity at the start of runoff may be larger than K_{sat}. Furthermore the K_{sat} measurements of the 0.5 m^2 plots were used at the scale of the flow units, without correcting for the decrease in runoff coefficient with area. The latter is documented by e.g. Rockström and Valentin (1997) and van de Giesen et al. (2000). Despite these limitations, the modelled amounts of runoff correspond well with other publications on semi-arid areas (Rockström and Valentin, 1997; Rockström et al., 1998; Gómez et al., 2004), which report annual runoff coefficients of 2.5-30%.

3.3.5 Implications of clean sweeping for tree transpiration

The absence of roots in the plough layer allows to split the soil water balance in two compartments, i.e. the plough layer and the subsoil. In principle only the water that drains below the plough layer can be taken up by the tree roots. Hence D_{15} (Eq. 3.1) is a direct estimate of the water availability for the trees. In practice the actual transpiration by the trees may be lower, because the soil below 15 cm can contribute to evaporation as well. Furthermore the roots may not be able to extract all water that infiltrates the subsoil. For example young trees have not yet developed an extensive root system. The roots of established trees are able to penetrate the entire subsoil depth over which D_{15} is distributed. They may extend more than 15 m from the trunk and up to 4 m deep (Catlin, 1996). The maximum depth of the wetting front simulated by BUDGET is reached at the sink position of C1. Two storms during the wet year (45.6 mm on day 272 and 32.8 mm on day 278, Fig. 3.8) produce a total drainage of 189 mm from the 35 cm profile in the sink. This water can percolate to a depth of 95 cm, assuming a field capacity of 0.32 m^3 m^{-3} and an initial moisture content of 0.13 m^3 m^{-3}. It therefore appears that soil water rarely percolates deeper than the potential rooting zone of the trees.

The occurrence of D_{15} is restricted to the period from September to April, as determined by the seasonal distribution of rainfall (Fig. 3.8). The question arises whether D_{15} is large enough to support the transpiration of the almond trees. The canopy cover lasts during the growing season (March - November) and is in the order of 3.2% as measured for C1. Under the assumption that the canopy transpires at 100% of ET_0 during the growing season, the orchard transpiration requirement is 28.5 mm in the wet year, 29.0 mm in the average year and 29.2 mm in the dry year. It appears that the transpiration requirement is considerably smaller than D_{15} in the average and wet year (Table 3.5). This is not the case in the dry year, when D_{15} = 0 mm, implying that the trees will suffer from severe water stress. Several arguments indicate that the trees may still survive in dry years. The BUDGET runs show that drainage below 10 cm in the dry year still is in the order of 16 mm, which equals 100% of ET_0 for a canopy cover of 1.7%. In addition, there may be some water stored in the subsoil if the preceding year was normal or wet. Furthermore, the water requirement of the trees can be reduced by pruning the canopy (Tubeileh et al., 2004). Another strategy is to abstain from tillage during dry periods, so that the tree roots can access the water stored in the top soil after light rainfall. Lastly, it can be noted that it is common in Murcia for a small fraction of the almond trees to die, see e.g. the gaps in the regular tree pattern in orchard C1 (Fig. 3.1).

The magnitude of E_p largely depends on the thickness of the plough layer. The 15 cm assumed in this study is in agreement with the field survey of roots and literature (Pastor, 2004). A management consisting of no-tillage combined with chemical weeding seems an interesting alternative for clean sweeping, because it enables the tree roots to grow into the plough layer and take up more water from light rains. Furthermore, no-tillage leads to a considerable decrease in soil erosion (Pastor, 2004). Additional experimental evidence is required to determine the effect of no-tillage on orchard productivity in dry areas. A drawback on silt loam soils might be an increase of runoff due to crust formation, so that minimum tillage (5-10 cm) is required in order to maintain infiltration rates (Pastor, 2004).

3.4 Conclusion

Rainfed orchards on silt loam soils in Murcia, southeast Spain are characterised by considerable evaporation losses. The practice of clean sweeping limits the water availability to the trees, since the root growth in the plough layer is constrained and drainage to deeper soil layers is limited. The spatial redistribution of runoff does not have a large influence on the water availability for the trees in the current cultivation system. An exception is formed by depressions in the landscape, where infiltration of run-on can significantly increase the water availability. The relative importance of the evaporation loss from the plough layer decreases with increasing rainfall. Therefore especially in dry areas a reduction of evaporation from the plough layer will contribute to an increased productivity and water use efficiency in rainfed orchards. The most promising techniques to achieve this are no-tillage combined with chemical weeding and minimum tillage (tillage depth limited to 5-10 cm). Both techniques decrease the thickness of the plough layer compared to the practice of clean sweeping and support the growth of tree roots close to the surface, where they can access the water from small rain events.

CHAPTER 4
PATTERNS OF WATER AVAILABILITY IN RAINFED ORCHARDS AT THE PROVINCE AND REGIONAL SCALE: VULNERABILITY AND POTENTIAL FOR COVER CROPS

4.1 Introduction

The Mediterranean region is recognised as a hot spot of soil erosion, which results from the interaction of climate, soil, topography and land use (e.g. Poesen & Hooke, 1997; Boardman, 2006). One of the most effective ways to decrease soil loss in agriculture is to increase the vegetation cover (Elwell & Stocking, 1976; Hartwig & Ammon, 2002). Successful examples include the growth of cover crops after the harvest of the main crop and the construction of grassed waterways (Fiener & Auerswald, 2003; Tonnito et al., 2006). One of the potential drawbacks for cover crops in semi-arid areas is the competition for water with the main crop (Ingels et al., 1998). Still little is known about the threshold conditions for the successful use of cover crops in terms of water availability. There are a number of studies that illustrate the effect of cover crops on the water balance (e.g. Pastor, 2004; Islam et al., 2006) and yields (e.g. Sánchez-Girón et al., 2004; Tonitto et al., 2006). Most of these studies address the application of cover crops from a plot-scale perspective. They offer neither general conclusions about the physical limits (i.e. climate and soil) for the application of cover crops, nor about how the application of cover crops can best be adapted to local conditions of water availability.

The focus of this chapter is on rainfed olive and almond plantations on sloping terrain, which feature a sparse vegetation cover and high erosion risk (Faulkner et al., 2003; Gómez et al., 2005). Olive (*Olea europea* L.) and almond (*Prunus dulcis* Mill.) are important crops that occupy a large part of the arable land in the Mediterranean. They can be grown under dry conditions, at an annual precipitation as low as 200 mm (Tubeileh et al., 2004). In Spain, the major part of the olives and almonds are grown under rainfed conditions, i.e. 18% of the olive orchards and 6% of the almond orchards were irrigated in 2003 (MAPA, 2004). Orchards can be adapted to drier conditions by reducing the canopy area and tree density (Navarro & Parra, 2004). A smaller canopy area decreases the water requirement, whereas a larger tree spacing allows the individual trees to mine a larger soil volume for water. We are not aware of any studies that investigated the relationship between water availability and these crop parameters.

The spatial variability in available water is determined among others by climate, soil properties and topography (Sala et al., 1988; Gómez-Plaza et al., 2001; Ramos & Mulligan, 2005; van Wesemael et al., 2006). We use the following parameters to characterise the orchard response to patterns of water availability at different scales: projected canopy cover, tree density and the spatial distribution of orchards. Canopy cover is an index of tree vigour (e.g. Gálvez et al., 2004) whereas tree density has been used to characterise the response of natural vegetation to climate (Caylor et al., 2003). In rainfed orchards, the tree density is adapted to the local climate as well, so that the soil volume and available soil moisture per tree is sufficient. This was illustrated by the water balance study in Chapter 3, which showed that the drainage below the plough layer over a large surface provided enough water to support 175 to 260 almond trees per hectare at an annual rainfall of 300 mm, and a projected canopy cover of 3 to 4% .

The aim of this chapter is to identify (1) spatial patterns of the occurrence of rainfed orchards at the regional scale related to lithology (i.e. surface geology) and (2) patterns of orchard tree density and canopy cover across the Mediterranean related to climate. These patterns can be used as a rapid evaluation of the vulnerability of the orchards to soil erosion and the possibility to apply a protective vegetation cover without compromising the water availability to the main crop.

4.2 Materials and methods

4.2.1 Selection and characteristics of almond and olive orchards

Olive and almond are common rainfed tree crops which are often grown on hillslopes with marginal soils. The cultivation of olive is widespread throughout the Mediterranean. Therefore olives were chosen as an indicator crop that allows studying spatial patterns related to climate at the scale of the Mediterranean. In addition, olives are easy to identify on satellite imagery and are widely documented. Both almond and olive cropping systems feature a sparse vegetation cover as well as the application of clean-sweeping (i.e. frequent tillage to prevent weeds to compete for water with the trees and to increase infiltration (Pastor, 2004). An important difference in terms of canopy cover is that olives are evergreen whereas almonds are deciduous. Furthermore, the major commercial almond cultivars are self-unfruitful and require cross-pollination to produce almonds (Hendricks, 1996).

4.2.2 The role of lithology

The pattern of rainfed orchards was studied in the province of Murcia, Spain, an area of 11 300 km². The climate is dry with a mean annual rainfall ranging from 180 mm up to 600 mm in the mountains, whereas the majority of the cultivated area receives less than 300 mm (Garcia & Hernández, 2004). This is still well within the Mediterranean climate range when compared to parts of Tunisia and Syria that feature rainfed agriculture under conditions with a considerably larger water deficit (e.g. Tubeileh et al., 2004). The mean annual air temperature ranges from 14.5 to 18.5 °C and the diurnal air temperature range is 11 to 12 °C (ATEAM database: New et al., 1999, 2000; Centro Regional de Estadística de Murcia).

It was investigated if there is a preference for rainfed orchards to be located on a specific parent material. Almond was selected as the study crop in Murcia because the area under almond cultivation is four times larger than the area of olives (CREM, 2004), making almond more suitable to study the effect of patterns in lithology. A lithological map with a resolution of 1 km^2 was produced by aggregating the 76 units of the 1:50 000 geological map of Murcia into 15 units with similar parent material and texture. An almond data set with a resolution of 1 ha was extracted from the land use map of the Ministerio de Agricultura, Pesca y Alimentación from 2001. Mixed land uses of almond with olive, cereals, etc. were excluded. The almonds covered 5.9% of the land in 2003, which equals about four times the area planted with olives (CREM, 2004). No distinction was made between rainfed and irrigated orchards, assuming that the irrigated orchards (13% of the total area of almond orchards in 2001; CREM, 2004) were located on similar lithologies as the rainfed orchards. The fraction of the province covered by each lithological unit was compared with the fraction of almond plantations that occurred on that unit. If the latter is higher, this points to a concentration of plantations on the lithological unit. The effective scale of the analysis was 1 km^2, as determined by the geological map.

A statistical test was done to determine if the number of 1 km^2 grid cells that contain almond plantations is significantly higher or lower on specific lithological units than the average of the entire region. It is based on stochastic variable $\underline{k}$, which represents the number of grid cells that contain almond orchards. $\underline{k} \sim \text{Binomial}(n, p)$, where n is the total number of grid cells and p the fraction of cells with almond plantations. For a specific lithological unit, the null-hypothesis is that p is the same as for the entire region. The expectation of $\underline{k}$ is then equal to p times n. The alternative hypothesis is tested for every lithological unit that k is higher or lower than the expectation of $\underline{k}$ under the null-hypothesis.

4.2.3 The role of climate

The relation between climate, canopy cover and tree density in olive orchards was investigated at the Mediterranean scale. Data on the canopy cover and tree density in rainfed olive orchards were obtained from field measurements in Spain, literature, satellite imagery and colleagues from different countries (Appendix 2). The mean annual air temperature at the sites ranged from 12 to 20 °C, the diurnal air temperature from 7 to 13 °C and the annual precipitation from 280 to 920 mm (New et al., 1999; Mitchell et al., 2002). In total 58 observations were collected. During the analysis no distinction was made based on plantation age, since it was unknown for most of the observations.

For each site the climate was characterised by the ratio between annual precipitation (P) and reference evapotranspiration (ET_0; Allen et al. 1998), both expressed in mm a^{-1} (Eq. 4.1).

$$HI = P / ET_0 \qquad (4.1)$$

This ratio is referred to as the aridity index (UNEP, 1997) or humidity index (Yin et al., 2005). We prefer the term humidity index (HI) since the index value increases with increasing humidity. ET_0 was calculated using the Hargreaves equation (Hargreaves & Samani, 1985; Allen et al., 1998; Eq. 4.2)

$$ET_0 = 0.0023(T_{mean} + 17.8)(T_{max} - T_{min})^{0.5} R_a \qquad (4.2)$$

where ET_0 = reference evapotranspiration (mm a^{-1}), T_{mean} = mean annual air temperature (°C), T_{max} = average daily maximum air temperature (°C), T_{min} = average daily minimum air temperature (°C) and R_a = extraterrestrial radiation (mm a^{-1}).

HI was calculated with data from the ATEAM terrestrial climate database, using monthly precipitation and monthly average air temperature for the period 1971-1995 (New et al., 1999; Mitchell et al., 2002). *HI* was also calculated for the eight driest years of the 1971-1995 period (HI_{dry}). The grid resolution of the air temperature data used was 0.5° and the resolution of the precipitation data 10'. The precipitation data was used at a finer resolution than the air temperature data because the *HI* is most sensitive to this parameter. In addition, the ATEAM database only enabled a 10' resolution for precipitation while the amount of rainfall observations (19 295 stations) was much higher than those of diurnal air temperature range (10 727 stations) (New et al., 1999; Mitchell et al., 2002). The maximum average error in mean monthly climatology of the ATEAM database as based on cross-validation was 35 mm for precipitation, 1.1 °C for mean air temperature and 2.1°C for diurnal air temperature range in our study area (New et al., 1999).

4.3 Results

4.3.1 Spatial patterns at the regional scale: the role of lithology

In total 73% of the almond orchards in Murcia is located on Quaternary deposits, marls, shales and schists (Table 4.1).

Table 4.1. The area of almond orchards per lithological unit in the province of Murcia.

Lithological units		Almond orchards		
	km²	Surface fraction	Deviation* (%)	
Quaternary	4462	0.079	-5	a
Marls	2072	0.102	23	a
Limestone & dolomite	2070	0.033	-60	a
Shale & schist	762	0.136	64	a
Conglomerate	692	0.097	17	a
Marble	507	0.129	55	a
Other	634	0.079	-5	b
Overall	11385	0.083	0	

* Deviation from the mean
a: $p < 0.0005$
b: $p < 0.07$

There is a marked concentration of orchards on marls, shales, schists and marble. Probably the soils on these units provide a higher water availability than those on limestone and dolomite. The soils on shales and schists are shallow and have a high rock fragment content which reduces evaporation losses (van Wesemael et al., 1996). This is confirmed by the water balance simulations in Chapter 3 and in van Wesemael et al. (2006), which point out that soil evaporation is considerably lower for stony soils on slates and phyllites than for marl soils. Another factor that probably plays a role is that the marls and shales are mouldable, which facilitates the planting of new trees and clean sweeping, which is the common form of tillage. The high concentration of orchards on marble is surprising (Table 4.1). However, a closer look at this unit reveals that it is a mix of mainly marble, shales and schists (personal communication with G. González). The marble outcrops in Murcia are closely associated with shales and schists, which can make up close to 50% of some of the marble unit grid cells. Almonds will not be planted on the marble because the rock is quite hard. Hence, the plantations of the marble unit are mostly located on the schists and shales. Furthermore, it can be noted that the marble unit features abrupt transitions between rock types and is not suitable for most other crops.

There is no simple explanation for the moderate concentration of almond orchards on conglomerate (Table 4.1). The properties of conglomerates are variable and their matrix can be hard or soft. From the maps it appears that most of the orchards on conglomerate are located near Lorca and Puerto Lumbreras in the south-west of Murcia province, where the conglomerates are relatively soft (Fig. 4.1). This part of the province also has large areas of schists and shales, which show the highest concentration of almond plantations (Table 4.1).

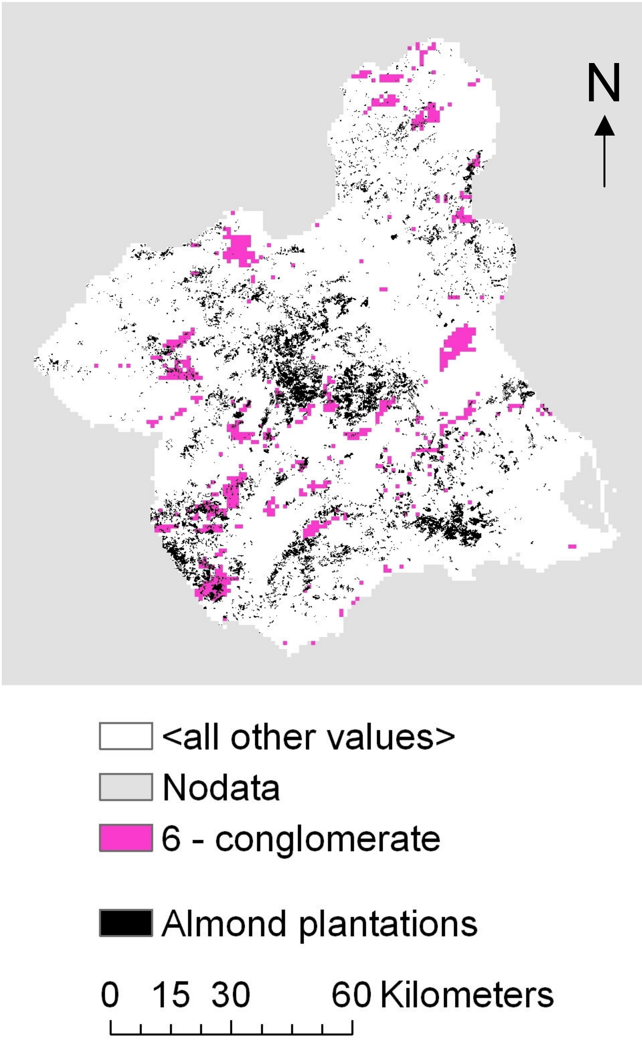

Figure 4.1. The spatial distribution of almond orchards and the lithological unit "conglomerate" in Murcia province. Note that the almond data set does not cover the north-east corner of the province.

The relative absence of almond orchards on limestone and dolomite is not surprising. The soils on these units are typically shallow and the soil pockets and cracks in the rock are not abundant enough to provide a deep storage of water for the roots (ICONA, 1986). Percolation is quick and soils on limestone are therefore relatively dry. The bedrock is not mouldable, which complicates the planting of trees. Furthermore, the landforms of limestone and dolomite in Murcia consist mainly of mountains and ridges with steep slopes, prohibiting the cultivation of crops. The only areas on these units that are suitable for agriculture are located in valleys and on pediments.

4.3.2 Spatial patterns across the Mediterranean: the role of climate

When considering the highest tree density observed at a given *HI*, there appears to be a linear increase in maximum tree density with *HI* for *HI* range 0.17-0.47 (Fig. 4.2a).

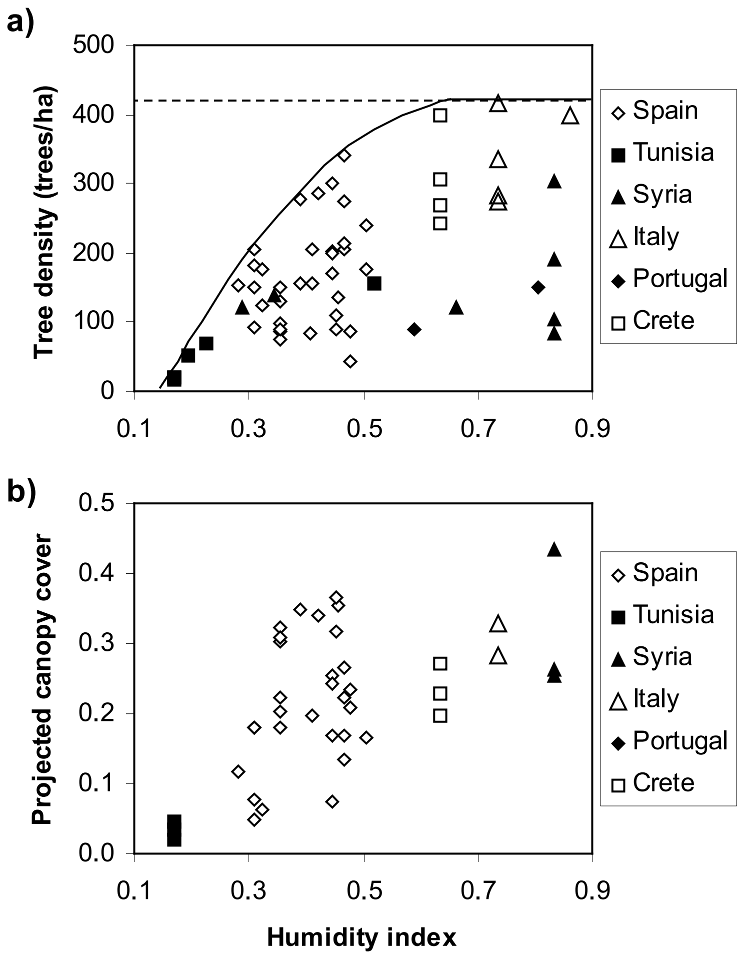

Figure 4.2. The relation between climate and (a) tree density ($r^2 = 0.320$, $n = 58$, $p < 0.001$) and (b) projected canopy cover ($r^2 = 0.319$, $n = 40$, $p < 0.001$) in rainfed olive orchards. The orchard properties and data sources are listed in Appendix 2. Climate data source: ATEAM database (Mitchell, 2002).

At HI > 0.60, the maximum tree density remains constant at approximately 400 trees ha^{-1}. This maximum can be related to two factors. Firstly, olive shoots require direct sunlight to produce flowers and fruits (Ferguson et al., 1994, pp. 31-38). Hence, higher densities may compromise the productivity because the sides of the trees become shaded. Secondly, mechanised management becomes less efficient when the distance between the rows is less than 7 m (Navarro & Parra, 2004). Canopy cover versus HI shows a similar behaviour, although somewhat less pronounced due to the lower amount of observations available (Fig. 4.2b). It seems that the scatter in canopy cover at a given HI is smaller than that of the tree density. This suggests that the orchard canopy cover is adjusted more consistently to climate than tree density. The correlations between the climate indices, canopy cover and tree density are summarised in Table 4.2.

Table 4.2. The correlation (r) between climate indices, orchard canopy cover and tree density. P = mean annual rainfall, HI = humidity index, the subscript "dry" refers to the average of the eight driest years from the period 1971-1995.

	Projected canopy cover	Tree density (trees ha^{-1})
P	0.58**	0.45**
P_{dry}	0.61**	0.44**
HI	0.57**	0.57**
HI_{dry}	0.60**	0.56**
Tree density (trees ha^{-1})	0.33*	-

* $p < 0.05$
** $p < 0.001$

Canopy cover shows a stronger correlation with mean annual rainfall than tree density ($r = 0.58$; $p < 0.001$). The observed relations are similar when considering only the eight driest years of the 1971-1995 period (Table 4.2).

It appears that mean annual rainfall is a better predictor of canopy cover than of tree density.

A considerable part of the variability in tree density and canopy cover that is not explained by climate probably results from differences related to plantation age, cultivars, soil conditions and regional cultivation practices. For example in south east Spain, a part of the olive orchards are old and more traditional with low tree densities and large trees, whereas the modern plantations are characterised by high densities and small trees (Beaufoy, 2003; Navarro & Parra, 2004). One important reason to apply higher tree densities is the increase in yield during the first years after planting compared to lower densities (Hendricks, 1996; Navarro & Parra, 2004).

Systematic differences in tree density between some regions in Spain are illustrated in Figure 4.3. In Murcia and west Jaén the tree density is somewhat higher than in south west Jaén, although the climate is more arid.

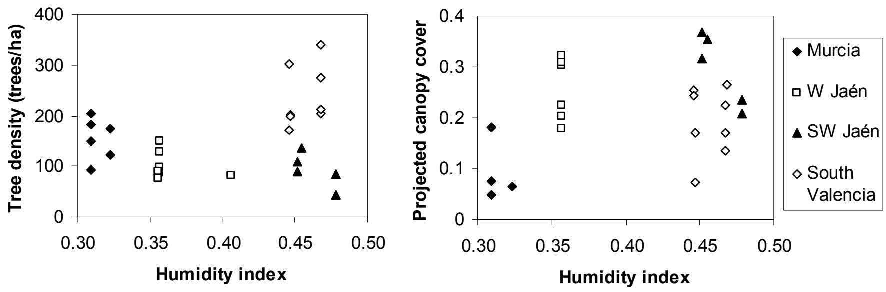

Figure 4.3. Regional differences in the tree density and canopy cover of rainfed olive orchards in Spain.

Traditionally, olive orchards in the more arid parts of Spain tended to be planted in cañadas (Giráldez et al., 1988). These are the bottoms of small valleys close to ephemeral channels, which benefit from runoff from the adjacent slopes and are often terraced. This may explain why the tree density in west Jaén and Murcia is relatively high compared to southwest Jaén (Fig. 4.3). The orchards in southwest Jaén are larger and feature big trees that are planted all over the landscape.

4.4 Discussion and synthesis

It is clear that patterns of water availability exist at the regional and Mediterranean scale and we attempt here to derive a number of (threshold) conditions and factors that can be used to assess if the growth of cover crops in a given location is possible without comprising the water availability to the trees.

It is hypothesised that the maximum tree density reached at $HI \geq 0.6$ reflects the point where the production is no longer limited by water availability (Fig. 4.2). This hypothesis can be strengthened by comparing our results with the crop water requirement as calculated using standardised FAO methodology (Allen et al., 1998). The crop water requirement for optimal growth is calculated (Eq. 4.3):

$$ET_c = K_c \, ET_0 \quad (4.3)$$

where ET_c is the crop water requirement (mm) and K_c a crop factor. In the case of optimal water availability in areas with water shortage, the annual precipitation (P) is equal to the crop water requirement (ET_c) and hence K_c is equal to HI (Eq. 4.1 & 4.3). The estimated K_c for olive of 0.6 from Figure 4.2 is close to K_c values reported in literature, which range from 0.55-0.75 (Allen et al., 1998; Villalobos et al., 2000; Ferguson et al., 1994, pp. 61-68). This means that at locations where $HI > 0.6$, there is a surplus of water that can be used to sustain the growth of a cover crop. This condition was observed in parts of Crete, Italy, Syria and Portugal (Fig. 4.2). Possibly, similar climatic thresholds can be defined for almond and other tree crops based on documented K_c values.

For olive, the existence of a digital olive tree database for Spain entitled "Sistema de Información Geográfica Oleícola Español" (available via the website of the Ministerio de Agricultura, Pesca y Alimentación) allows to compile tree density data for the whole of Spain. When this data is combined with a climate database like ATEAM, the relations identified in this study may be further explored. In addition, a link between the Oleícola database and harvest data will enable to determine the tree spacing for optimal production as a function of climate (*HI*).

Below the climatic threshold ($HI < 0.6$ for olive), the application of cover crops must be restricted to avoid competition for water between the cover crop and the trees. Under these conditions, the total canopy cover is clearly limited by climate (Fig. 4.2). It can be considered to grow cover crops during only a part of the season, i.e. the winter period, and on only a part of the area, i.e. in zones of concentrated flow (e.g. Fiener & Auerswald, 2003; Pastor et al., 2004). Depending on spatial patterns of water availability at the province and field scale, specific locations with a higher water availability may still enable the growth of cover crops. This is illustrated by the results of §4.3.1, which show that the water availability at the province scale is partly related to patterns in lithology. The Quaternary deposits and marls can provide a larger rootable soil volume than limestone, dolomite and marble. Soils on shale and schist are shallow and stony but friable, so that roots can penetrate in the weathered bed rock. Furthermore, due to the low water retention capacity of the stony soils, rainfall penetrates to a greater depth where it is protected against evaporation. The presence of a stone mulch in valleys further reduces the water loss by evaporation (van Wesemael et al., 2000; Li, 2003). The presence of stone mulches is promoted by the regular tillage of the orchards (Poesen et al., 1997; Oostwoud Wijdenes et al., 1997).

Our observations point out that typically more than 60% of the soil surface in olive and almond orchards in southeast Spain is bare (no canopy or weed cover). Hence, the reduction in evaporation and the rapid infiltration associated with the stone mulch on shale and schist soils have an important positive effect on the water availability to the crop (Li, 2003; van Wesemael et al. 2006). Additional research is required to quantify to what extent the roots of the almond trees are able to benefit from the water availability underneath the plough layer.

The previous discussion indicates that there are still opportunities for the growth of cover crops at locations that are below the climatic threshold. Depending on the rainfall regime, water for cover crops may be available in zones of runoff concentration and thalwegs. Under these circumstances, grassed waterways form an effective means of erosion control (Fiener & Auerswald, 2003). The implementation of grassed waterways can even be considered if this results in competition for water with the tree crop: the thalwegs and grass strips cover a relatively small part of the field and hence the overall effect on tree vigour and yields remains limited.

4.5 Conclusion

At the province scale in Murcia, southeast Spain, there is a preference for almond orchards to be located on stony soils on shales and schists and non-stony soils on marls. This indicates that the spatial patterns in lithology and soil properties can make a difference in terms of water availability and the growth of cover crops. Stony soils feature low evaporation and runoff losses compared to non-stony soils, since they have a low water holding capacity which promotes percolation to the subsoil. The mouldable parent material of marl deposits facilitates site preparation, planting and tillage of the orchards. The occurrence of almond orchards on limestone and dolomite is low because these rocks are not easily mouldable and their steep slopes are unsuitable for agriculture.

The spatial variability in available water has important consequences for the growth of cover crops in rainfed orchards. The upper limit of available water and canopy cover in olive orchards is determined by climate, as expressed by the ratio of mean annual precipitation and reference evapotranspiration (*HI*). For $HI < 0.5$ this limit is described by a simple linear relation between maximum tree density and *HI*. The olive canopy cover reaches a maximum around $HI = 0.6$. At $HI > 0.6$ there is a surplus of water that could be used to grow cover crops without compromising the water availability to the trees. Below this threshold value, the growth of cover crops is possible in locations with a relatively high water availability as determined by spatial patterns in the landscape. Furthermore, the application of cover crops can be adjusted to drier conditions by killing them off in spring.

The results of this study may serve as a basis for further investigations on increasing the vegetative cover in rainfed orchards. In recent literature this topic has received little attention, despite the fact that the major part of the olives and almonds are grown under rainfed conditions. Combination of the Oleícola olive tree database with yield records and the ATEAM climate database will enable a more elaborate assessment of the opportunities for cover crops in Spain.

CHAPTER 5
THE IMPACT OF TERRACE FAILURE ON HYDROLOGICAL CONNECTIVITY AT THE CATCHMENT SCALE

This chapter has been submitted for publication in Hydrological Processes: Meerkerk, A.L., van Wesemael, B., Bellin, N. Overland flow in semi-arid catchments: the impact of terrace failure on hydrological connectivity.

5.1 Introduction

Agricultural terraces form an important feature of sloping landscapes in the Mediterranean and other semi-arid areas (Grove and Rackham, 2003). Most have been constructed for agricultural purposes, and some of the older terraces have been dated to c. 500 BC. The purpose of terracing and its effect on hydrological processes depends on geology and soil properties (Grove and Rackham, 2003). Terraces reduce the slope gradient and length, facilitating the cultivation of crops on steep slopes. Furthermore, terraces increase the infiltration of rainfall and run-on in areas with a moderate to low soil permeability, which is commonly referred to as water harvesting (Ennabli, 1993; van Wesemael et al., 1998; Yuan et al., 2003).

Despite the purpose and benefits perceived by the farmers who built them, there has been a prolonged trend of abandonment and removal of terraces in the Mediterranean, which goes back at least a hundred years (Grove and Rackham, 2003). In many cases the maintenance of terraces has ceased following land abandonment and in other cases terraces have been removed to facilitate the mechanisation of cropping systems. As a result of terrace failure and removal, a larger part of the landscape becomes connected to the river by continuous pathways of concentrated overland flow during storms. This implies that the discharge of ephemeral rivers in agricultural landscapes may have increased over the past century. In the case of land abandonment, the increase in connectivity may be mitigated by a recovery of natural vegetation (Cerdà, 1997; Ruecker et al., 1998). In the case of continued cultivation it is likely that the probability of high peak discharges and the flooding of downstream residential areas increases following terrace removal. Other possible effects include an increase in sediment fluxes (Ruecker et al., 1998; Koulouri & Giourga, 2007).

The concept of hydrological connectivity is a key element of our understanding of rainfall-runoff processes in semi-arid catchments (Cammeraat, 2004; Croke et al., 2005; Ocampo et al., 2006; Mueller et al., 2007; Reaney et al., 2007). The presence and configuration of connected features like pathways of concentrated flow and saturated thalwegs can strongly affect discharge and sediment transport during storms (Western et al., 2001; Cammeraat, 2004; Croke et al., 2005; Reaney et al., 2007). In general, river discharge and sediment transport increase with increasing connectivity, which is not captured by more conventional rainfall-runoff models (Western et al., 2001; Bracken & Croke, 2007; Mueller et al., 2007; Reaney et al., 2007; Verstraeten et al. 2007).

Despite a considerable number of studies that use or address the concept, there is no broad consensus on the definition of hydrological connectivity (Bracken & Croke, 2007) and quantitative measures are not well established.

This may result from the fact that connectivity studies are often case-specific, limited to particular land units and scales, and focused on the flux of different elements like water, sediment and fauna (Amoros & Bornette, 2002; Hooke, 2003; Pringle, 2003; Stieglitz et al., 2003; Croke et al., 2005). Different terminology similar to hydrological connectivity is used, including "topographic connectivity" (Western et al., 2001), "fluvial coupling" (Harvey, 2002), and "scale dependent thresholds" (Cammeraat, 2004). A comprehensive review on the present use of the connectivity concept is provided by Bracken & Croke (2007). They mention two issues that are addressed in this chapter: the knowledge gap on the relation between storm properties and hydrological connectivity, and the need for a quantitative variable that can be measured in the field. Few studies are available that address hydrological connectivity at the catchment scale and that present field data on pathways of overland flow (Western et al., 2001). It appears that most recent field studies have focused on the connectivity between road networks and the river system (Wemple et al. 1996; MacDonald et al., 2001; Arnáez et al., 2004; Croke et al., 2005).

The work of Western et al. (2001) in our opinion comes closest to a general quantitative definition of hydrological connectivity. The use of connectivity functions and the integral connectivity scale appears to be a promising way to quantify hydrological connectivity (Allard, 1994; Western et al., 2001). Connectivity functions provide a probabilistic model of connectivity based on observed patterns of e.g. soil moisture and concentrated overland flow. An advantage of connectivity functions is that they are continuous in space and based on geostatistical principles. They can be determined in one to three dimensional space and for different points in time.

In the context of overland flow we introduce a connectivity function, which we refer to as the contributing area function. This function reflects the extent and spatial configuration of the contributing area of overland flow and enables to demonstrate the link between hydrological connectivity and contributing area.

This chapter focuses on an agricultural catchment in south-east Spain, for which detailed data were gathered on the presence of terraces and small dams in 1956 and 2005 (Bellin et al., submitted). The chapter aims (1) to quantify the hydrological connectivity by concentrated overland flow; (2) to determine the effect of terrace removal and failure on hydrological connectivity and peak discharge, based on the state of the terraces in 1956, 2006 and a scenario where all terraces have become permeable for concentrated flow; (3) to determine the control of storm parameters, land use and topography on hydrological connectivity and the occurrence of concentrated overland flow.

5.2 Materials and methods

5.2.1 Study area

The catchment is part of the Cárcavo basin and is located in the region of Murcia, Spain (38.20° N, 1.52° W). The extended subcatchment measures 475 ha (Fig. 5.1). A southern (180 ha) and southeastern (130 ha) subcatchment are distinguished because of the limited extent of some of the data sets and analyses (Fig. 5.1).

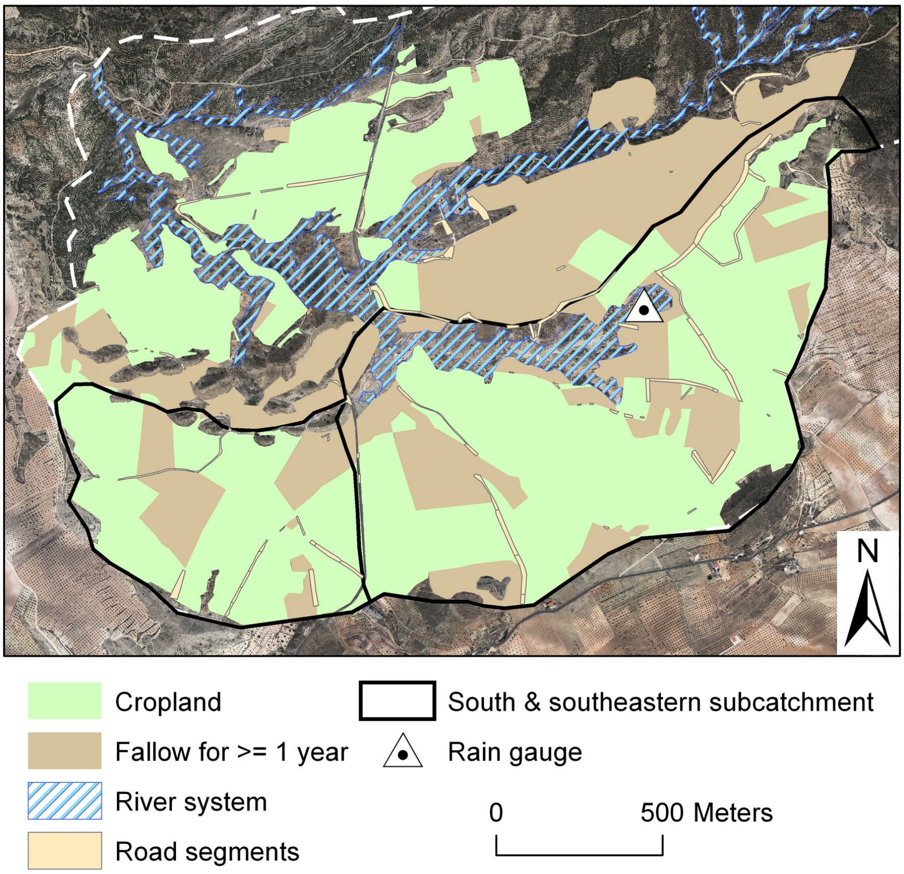

Figure 5.1. Study area and land use. The contour of the extended subcatchment is indicated by the dotted white line.

The annual rainfall is 298 mm and the mean annual air temperature is 16.4° C (Almadenes station, period 1966-2000, 38.23° N, 1.55° W). The climate is characterised by a high interannual rainfall variability and a potential evapotranspiration of 1125 mm y^{-1} (Chapter 3). The geology consists mainly of Cretaceous marls and Quaternary colluvial and alluvial deposits. Furthermore, there are Cretaceous limestone and sandstone outcrops, Triassic dolomites and gypsiferous marls, which are mostly confined to the north and south of the extended subcatchment (Fig. 5.1). The agricultural fields are concentrated on the marls and Quaternary deposits. Their soils are classified as Calcaric Regosols with a silt loam texture and an organic matter content of c. 6 g kg^{-1} (ICONA, 1986). Granulometry and hydraulic properties are provided in Chapter 3. Additional information on the Cárcavo basin has been published by Castillo et al. (2007) and Lesschen et al. (2007).

The crops cultivated are almond, olive and grapevine. A part of the agricultural land has been abandoned during the last decades, as documented by Lesschen et al. (2007). This trend continued during the 2004-2007 field work period, when some cereal fields were allocated for hunting and a number of almond fields were no longer ploughed nor harvested. For the application in this chapter we divide the agricultural area in two classes based on the situation in January 2006. The class "cropland" is the area where crops were actually grown and "fallow land" is the area that had not been cultivated or ploughed for at least one year (Fig. 5.1).

The conservation structures in the area consist of terraces and check-dams (Fig. 5.2). In general, they are not re-enforced with stones and do not have a spillway. The dams are less than two meters high and occur in thalwegs with a gentle plan curvature. The terraces are on slopes and in narrow thalwegs. The structures can be classified as "step terrace", "braided terrace" and "check-dam terrace" according to the typology of Grove and Rackham (2003). Most terrace risers are covered by weeds, depending on age and stability.

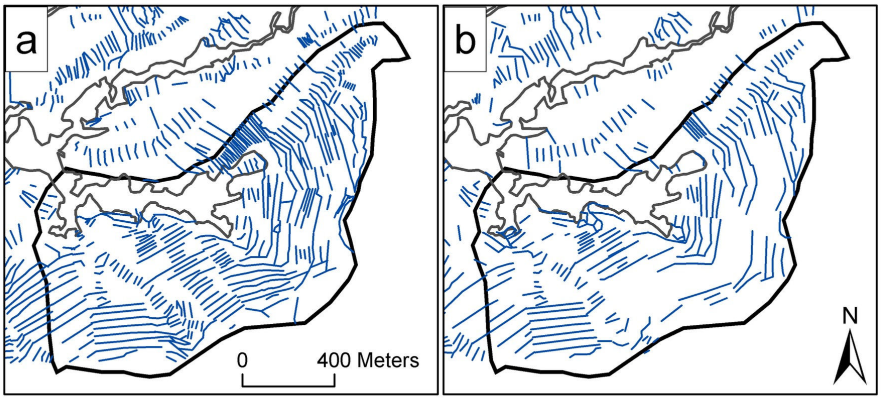

Figure 5.2. Terraces and check-dams in 1956 (a). Terraces and check-dams that were impermeable to overland flow in 2006 (b).

5.2.2 Survey and analysis of concentrated flow

Traces of concentrated overland flow were mapped after four storms in 2006 in the agricultural area (Fig. 5.1). The observed traces include rills and sediment deposits on fields and dirt roads, erosion and sedimentation in existing gullies, new gullies, pipes, signs of flow such as bent grass, mud drapes after ponding of water on terraces and in depressions. In order to carry out the surveys as quickly as possible two GPS systems were used at the same time: a small hand held GPS (Garmin) with a 95% positional accuracy of 10 m and a more sophisticated GPS (Trimble TSCe) with a 95% accuracy of 1 m. The time lag between the events and the field survey was 6 days for event E1, 58 days for E2, 27 days for E3 and 3 days for E4. Rainfall was measured at a one minute interval using a tipping bucket rain gauge

(Fig. 5.1). The return period of the storms was determined from the 35 years of rainfall data from Almadenes station. Individual events were distinguished if dry spells lasted more than five hours.

The control of storm properties, land use and topography on the occurrence of concentrated flow was characterised for the southern subcatchment by means of scatter plots and regressions. One of the independent variables was the storm erosivity index EI_{30}, which is defined as the product of the total kinetic energy of the storm and the 30 minute peak intensity of the storm (Agnese et al., 2006). The energy term of the index was calculated from the rainfall measurements (1 minute resolution) using the general equation of van Dijk et al. (2002), which was adjusted for the mean elevation of the catchment (Eq. 5.1:)

$$E_{\mathrm{k}} = 29.48\left(1-0.52\,e^{-0.042R}\right) \qquad (5.1)$$

Where E_{k} is rainfall kinetic energy (J m^{-2} mm^{-1}) and R rainfall intensity (mm h^{-1}).

Three classes of land use were distinguished: cropland, fallow land, dirt roads. The cereals fields were fallow in 2006 so that the class cropland consists of perennial crops. These crops all feature a low vegetation cover (typically less than 10%) and regular tillage (Chapter 3). A digital elevation model (DEM) was interpolated from the 1:5000 topographical map using the topogrid interpolation technique (Hutchinson, 1989). A cell size of 10 m was chosen based on the 5 m contour interval of the topographic map. The flow and land use data sets were converted to 10 m rasters as well. Flow paths shorter than 10 m were excluded so that subsequent analyses would be consistent in terms of scale. The working hypothesis was that the contributing area of runoff equals the area of 10 m grid cells through which the observed flow lines run. This area is subsequently referred to as the area producing concentrated flow (A_Q). In order to determine the distribution of A_Q over the land use classes, a buffer of $5\sqrt{2}$ m was applied to the digitised road segments to ensure that all A_Q cells originating from flow lines on roads would be attributed to roads. Without the buffer, a large part of these cells would be classified incorrectly and considered off-road, whereas their centre lies outside the road segments, which are only a few meters wide.

5.2.3 Quantification of connectivity

In quantitative terms, connectivity is a binary variable defined for two points separated by distance *h*. Whether or not the two points are connected can be determined based on a binary pattern that is derived from spatial variable *v* (Allard, 1994; Western et al., 2001). In the case of hydrological connectivity, *v* may consist of pathways of flowing water, saturated thalwegs, soil moisture, etc. A binary field *Z* is produced by thresholding *v*, so that *Z* consists of pixels with value 0 or 1 (Western et al., 2001). Let *F* be the set of pixels with $Z = 1$. A pair of pixels is considered to be connected if both pixels belong to *F* and if there is a continuous pathway of pixels between them that also belong to *F*. The degree of connectivity in field *Z* is defined as the probability that a pair of pixels separated by distance *h* is connected. This is expressed by the connectivity function $\tau(h)$ (Eq. 5.2, adapted from Western et al., 2001):

$$\tau(h) = P(x \leftrightarrow x + h) \qquad (5.2)$$

The connectivity function of Eq. 5.2 is different compared to the function of Western et al. (2001). The difference is that in Eq. 5.2, $\tau(h)$ is expressed as a fraction of the pixels in Z, whereas in the equation of Western et al. (2001), $\tau(h)$ is expressed as a fraction of the pixels in F. Eq. 5.2 was used here to enable a more straightforward comparison between $\tau(h)$ graphs of different patterns. When $\tau(h)$ is scaled to F, then by definition $\tau(0) = 1$. When $\tau(h)$ is scaled to Z, $\tau(0)$ equals the fraction of pixels in Z that are part of F. Hence, $\tau(h)$ graphs produced with Eq. 5.2 not only provide the connectivity as a function of h, but also show the fraction of cells in Z with $Z = 1$.

It should be reminded that the term connectivity may refer both to the binary variable (two objects being connected or not) and to the degree of connectivity in field Z as a whole, which contains many objects. This may in part explain the confusion on the term connectivity in the literature (Bracken & Croke, 2007). The integral of the $\tau(h)$ function, denoted by I_τ, is defined as the integral connectivity scale and represents the average distance over which grid cells are connected (Western et al., 2001; Eq. 5.3):

$$I_\tau = \int_0^\infty \tau(h)\, dh \qquad (5.3)$$

$\tau(h)$ can be referred to as the omnidirectional connectivity, which means that any pathway that connects the pair $(x, x + h)$ is considered valid, irrespective of its trajectory or length. In the case of hydrological connectivity, a pair is defined connected only if there is a valid hydrological pathway between them, i.e. a pathway that is followed by a flow of water. We denote the hydrological connectivity function as $\psi(h)$ to distinguish it from $\tau(h)$.

The contributing area of flow can be expressed as a connectivity function as well, denoted by $\alpha(h)$. This function does not consider connectivity between random pairs of points like $\psi(h)$, but the connectivity between a point and a specific object like a catchment outlet or a permanent drainage line. The specific object can consist of a point, line or surface, or a combination of these. Let F be the set of pixels in binary field Z with $Z = 1$, let G be the set of pixels in Z outside the specific object and H the set of pixels in Z that make up the edge of the specific object. The contributing area can now be defined as the probability that a pixel in G separated by distance h from H is connected to H. Distance h is defined here as the minimum Euclidian distance between x and H. Pixel x is considered to be connected to H if x belongs to F and if there is a continuous and hydrologically valid pathway of pixels between x and H that belong to F as well. The contributing area function $\alpha(h)$ is expressed as (Eq. 5.4):

$$\alpha(h) = P(x \leftrightarrow x + h \mid x \in G, x + h \in H) \qquad (5.4)$$

More simply put, $\alpha(h)$ is the probability that a pixel x at distance h from a specific object is connected to it.

Programs were written in C++ to implement the calculation of $\tau(h)$, $\psi(h)$ and $\alpha(h)$. This was based on the pseudocode provided for $\tau(h)$ in Western et al. (2001). The $\psi(h)$ algorithm uses a DEM as additional input, which was derived from the 1:5000 topographical map (§5.2.2). Starting at the upslope pixel of a pair of cells $(x, x + h)$ in F, the $\psi(h)$ algorithm considers adjacent pixels as connected if they belong to F and if their elevation is equal to or lower than that of the upslope pixel (i.e. overland flow cannot flow in upslope direction). Next, this rule is applied to all connected downslope pixels until the entire pathway has been detected. If the downslope pixel of the pair is part of the detected pathway, then the pair is hydrologically connected. The algorithm is similar to that for "topographic connectivity" of Western et al. (2001).

In order to calculate $\alpha(h)$, the $\psi(h)$ algorithm was modified so that it only considers pairs of which at least one pixel is part of the edge of the specific object. Similar to I_τ, the integral connectivity scale of $\psi(h)$ and $\alpha(h)$ is denoted as I_ψ and I_α. The connectivity functions were calculated for the observed pathways of concentrated flow and a number of scenarios (Fig. 5.3 & 5.4). The analysis was confined to the southeastern subcatchment because of the limited extent of the data set of event E1 (Fig. 5.1 & 5.3).

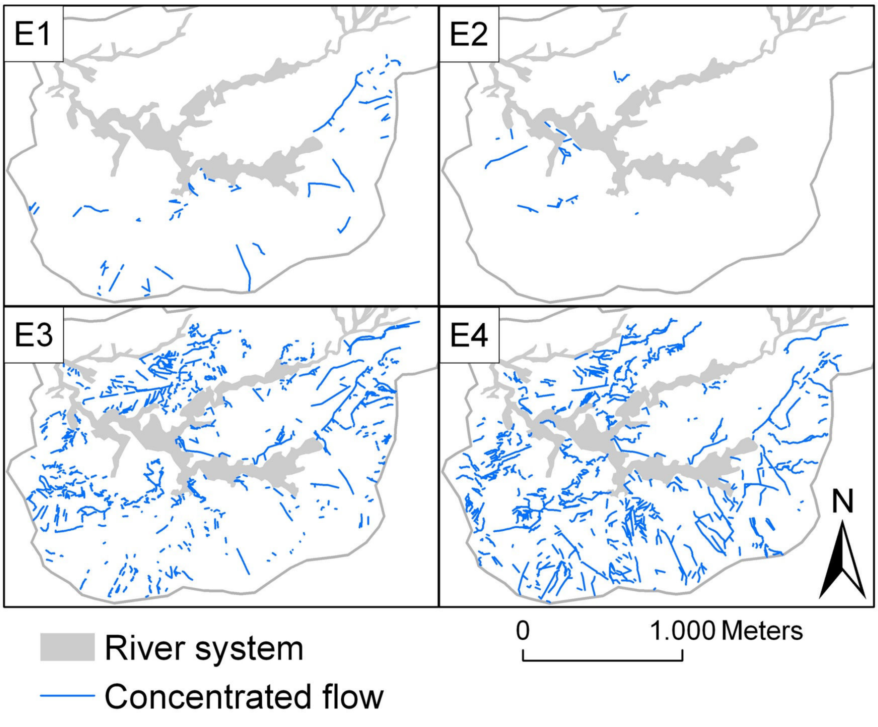

Figure 5.3. Pathways of concentrated flow after storms in 2006. The storm properties are listed in Table 5.1. Note: the E1 data set only covers the southern subcatchment (Fig. 5.1).

Table 5.1. Storm properties and area producing concentrated flow (A_Q) observed in the agricultural area of the southern subcatchment in 2006.

Event	P [a] (mm)	I_{10} [b] (mm h^{-1})	EI_{30} [c] (MJ mm ha^{-1} h^{-1})	T [d] (y)	A_Q (ha)
E1 (January)	25.9	19.2	82.1	0.50	5.0
E2 (April)	11.4	30.0	35.8	0.12	0.2
E3 (September)	23.2	67.5	222	0.38	12.1
E4 (November)	92.8	55.8	709	8.2	23.3

[a] Precipitation sum; [b] Ten minute peak intensity; [c] Storm erosivity index (Agnese et al., 2006); [d] Return period of the event.

5.2.4 Scenarios of concentrated flow

Three scenarios of concentrated flow were defined based on storm E4, in order to examine the impact of a decline of terraces on hydrological connectivity (Fig. 5.2 & 5.4, Table 5.1). The first scenario represents the situation in 1956 (denoted by S1), for which it is assumed that the concentrated flow observed for event E4 is stopped if it arrives at a terrace or check-dam that was present in 1956 (Fig. 5.2 & 5.4). The second scenario (S2) represents a situation without terraces. For S2 it is assumed that the flow lines observed for event E4 that enter a thalweg with an upslope drainage area ≥ 1 ha are not stopped by any barrier and run through the thalweg into the river. Furthermore, it is assumed that the land use of S1 and S2 is the same as in 2006 (E4) so that the presence of impermeable terraces forms the only difference. A third scenario (S0) consists of a random distribution of concentrated flow, where the number of flow pixels is equal to that of E4.

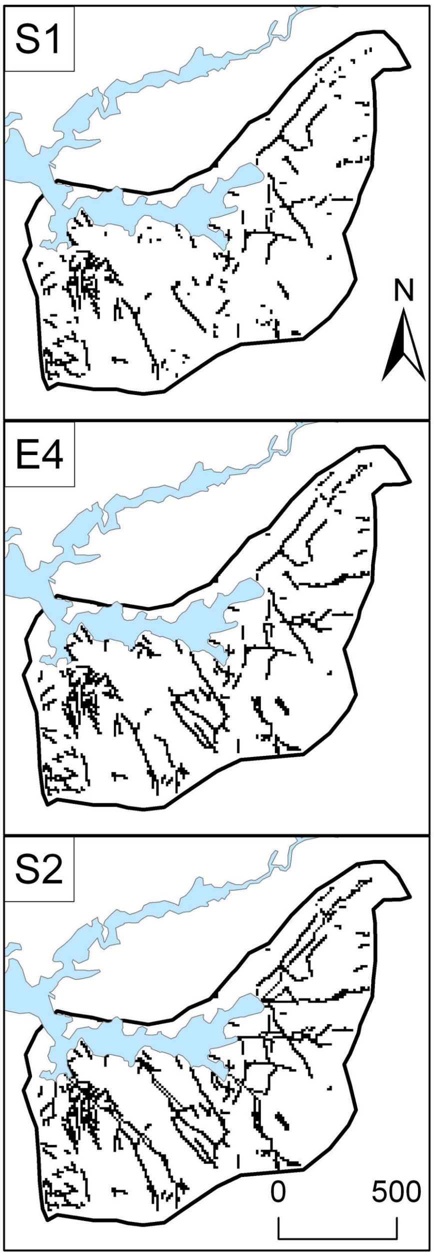

Figure 5.4. Pathways of concentrated flow generated by event E4 for different scenarios of terrace failure and removal. S1: terraces of 1956 assumed intact (Fig. 5.2a); E4: situation in 2006 (Fig. 5.2b); S2: situation without terraces. The scale bar is in meters.

5.2.5 Estimation of runoff and discharge

The peak discharge from the cropland into the river was estimated using the rational method (Shaw, 1994, Eq. 5.5).

$$Q_p = 0.278\ C\ i\ A \qquad (5.5)$$

where Q_p is peak discharge ($m^3\ s^{-1}$), C is the coefficient of runoff, i is the intensity of rainfall (mm h^{-1}) during time T_c and A is the area of the catchment in km^2. A was replaced by A_c, i.e. the area of concentrated flow connected to the river. T_c is the time of concentration, which is the time required for runoff produced at the most remote point to flow to the river. T_c (minutes) was estimated with the Kirpich equation (Kirpich, 1940; Eq. 5.6):

$$T_c = 0.015\left(\frac{L}{\sqrt{S}}\right)^{0.80} \qquad (5.6)$$

Where L is the length of the longest flow path to the river (m) and S the average slope gradient of that flow path. C (Eq. 5.5) was derived for period [0 , T_c] after the rainfall peak of the event, using the tipping bucket rain gauge data and infiltration measurements (Fig. 5.1; Chapter 3). The runoff production of the pixels connected to the river was calculated as a weighted average based on the number of connected pixels on cropland and fallow. It was assumed that runoff occurs after the initial abstraction has been reached, and infiltration capacity was assumed to be equal to the saturated hydraulic conductivity (K_{sat}). For cropland, K_{sat} was set at 26.1 mm h^{-1} and for fallow this was 19.3 mm h^{-1}. The initial abstraction was set at 3.8 mm for fallow and 6.5 mm for cropland. These are average values obtained from rainfall simulations in the catchment (Chapter 3). The same figures were used for all events, so that the runoff estimates do not account for differences in initial soil moisture.

5.3 Results and discussion

5.3.1 Storms and concentrated flow

The first three storms have a return period of 0.5 year or less (Table 5.1). Depending on the definition of the duration of a rain event, the return period of E4 varies from 2 to 23 years (Hooke & Sandercock, 2007). The figure of 8.2 years in Table 5.1 is obtained when individual events are distinguished if dry spells last more than five hours. The area producing concentrated flow (A_Q) is small in general and reaches a maximum of 23.3 ha for E4 or 15% of the agricultural area (Table 5.1 & Fig. 5.3). Event E2 generates little concentrated flow because the rainfall sum (11.4 mm) and peak intensity (I_{10} = 30.0 mm h^{-1}) are within the observed range of initial abstraction (3 to 14 mm) and K_{sat} (16 to 36 mm h^{-1}; Chapter 3). Therefore, only some parts of the catchment produced runoff and concentrated flow during E2. The rainfall depth of E1 and E3 is roughly twice that of E2. The ten minute peak intensity of E3 is 3.5 times higher than that of E1 and the storm energy (EI_{30}) and area with concentrated flow are about 2.5 times larger (Table 5.1 & Fig. 5.3). This illustrates the importance of the within storm rainfall distribution for runoff production in semi-arid areas (Nicolau et al., 1996; Puigdefabregas et al., 1998; Castillo et al., 2003; Reaney et al., 2007). Event E4 was relatively large, and resulted in a number of terrace failures (Bellin et al., submitted). When taking the four events of Fig. 5.3 together, 90% of the flow lines consist of rills, 8% of gullies and 2% of other flow traces.

5.3.2 Hydrological connectivity

The difference in the degree and maximum distance of hydrological connectivity between the events and scenarios is illustrated in Fig. 5.5. The shape of the connectivity functions is quite constant and typical for a random pattern (Western et al., 2001). However, the random appearance of our functions results from the seemingly random distribution of entire flow lines rather than the random distribution of the individual pixels with concentrated flow (Fig. 5.3). The difference is illustrated by comparing E4 to S0, which have the same number of flow pixels (Fig. 5.5). The pattern of E4 shows a higher degree of organisation and higher $\psi(h)$ values because the flow pixels are organised in randomly distributed lines, whereas the S0 pattern is entirely random, without distinct line elements. Note that $\psi(0)$ equals the area fraction of concentrated flow (§5.2.3).

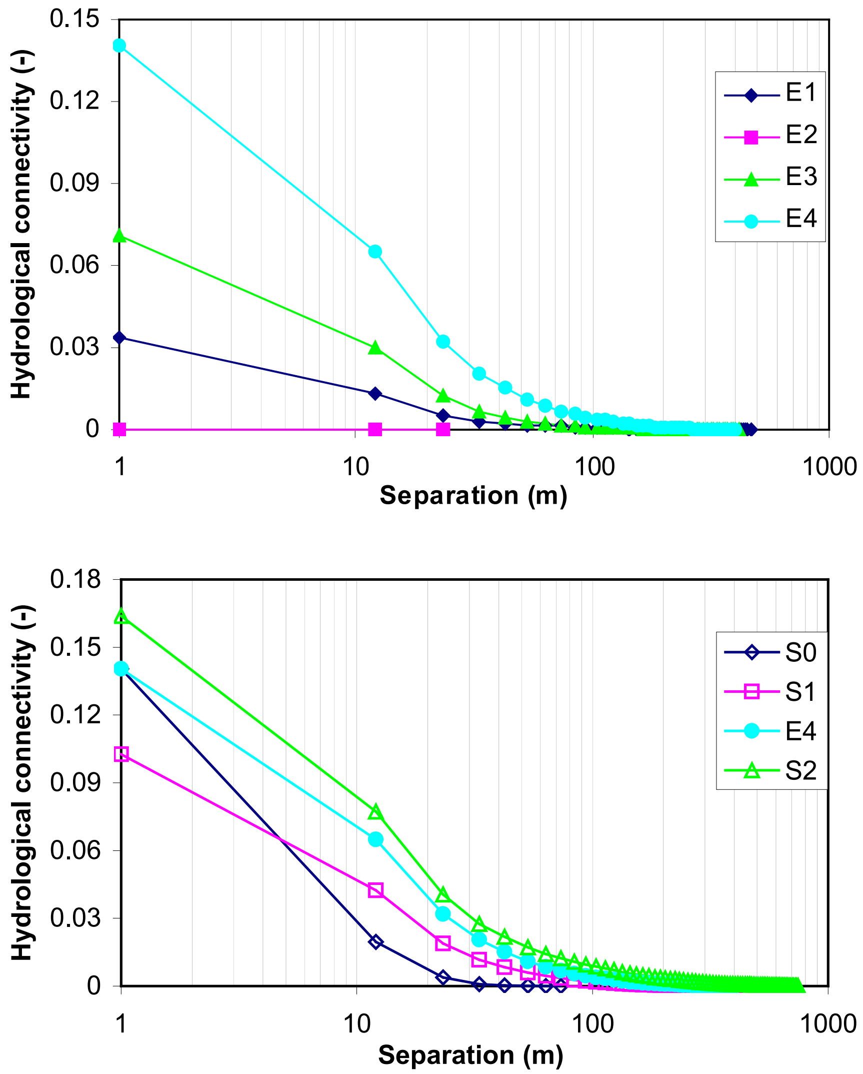

Figure 5.5. Hydrological connectivity function $\psi(h)$ of concentrated flow for a number of storms (Table 5.1). The input patterns are shown in Fig. 5.3 and 5.4 except for S0, which is a random pattern with the same number of flow pixels as E4. The data extent is limited to the southeastern subcatchment (Fig. 5.1). Note: $\psi(0)$ is plotted at $h = 1$ m.

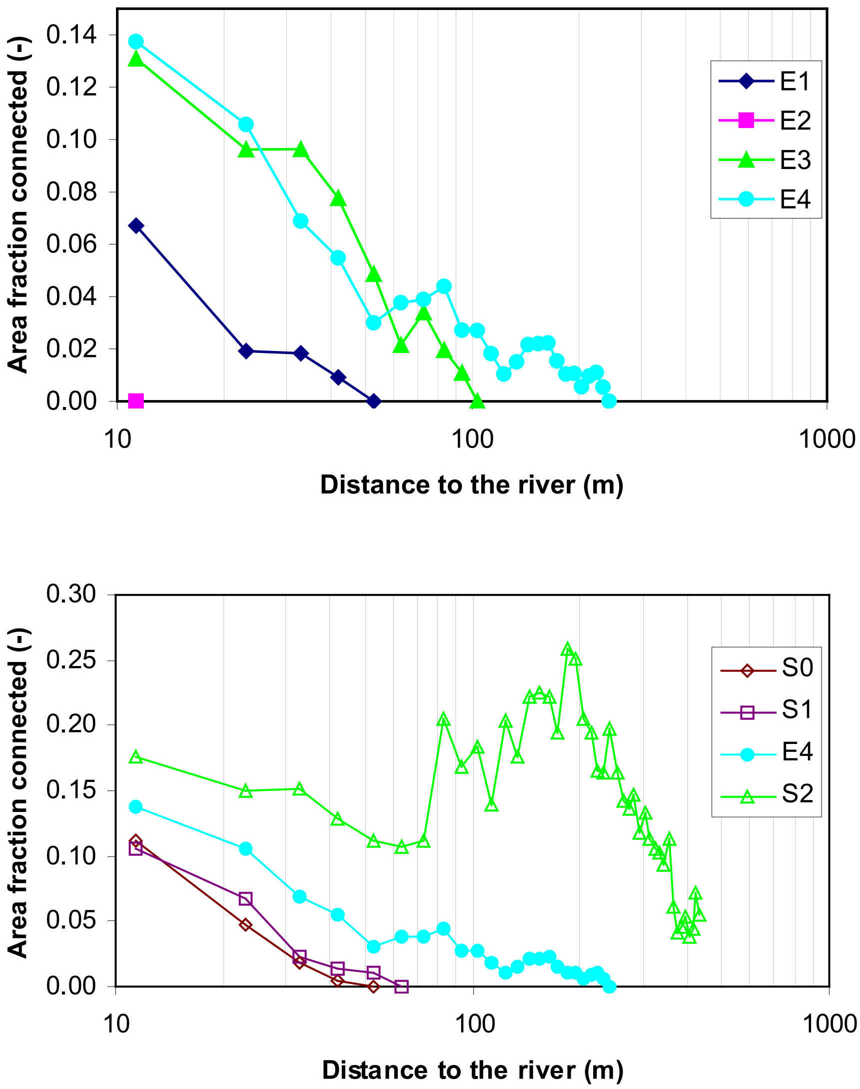

Figure 5.6. Contributing area function $\alpha(h)$ for a number of storms (Table 5.1). The input patterns are from Fig. 5.3 and 5.4, except for S0, which is a random pattern with the same amount of flow pixels as E4. The data extent is limited to the southeastern subcatchment (Fig. 5.1). The graph of S2 is cut off at h = 440 m, since the number of pairs per 10 m lag bin decreases quickly at larger h and the corresponding $\alpha(h)$ values are unreliable.

Whereas the $\psi(h)$ functions capture the overall degree of hydrological connectivity, the $\alpha(h)$ functions illustrate the effect of storm properties and the configuration of terraces on the contributing area of concentrated flow (Fig. 5.6), which is obtained by summing up $\alpha(h)$ times the area of the catchment in lag bin h, for all values of h. Hence, $\alpha(h)$ represents the contributing area of concentrated flow as a function of the distance to the river.

The degree of connectivity between the agricultural land and the river ($\alpha(h)$) is considerably larger than the degree of connectivity within the agricultural land itself ($\psi(h)$) at a given lag distance (Fig. 5.5 & 5.6). This is not surprising as the overall topography induces a convergence of flow lines to the river (Fig. 5.4). The integral connectivity scale further illustrates that the degree of connectivity to the river is larger than the overall degree of hydrological connectivity, i.e. I_α is consistently larger than I_ψ (Table 5.2, next §).

The $\alpha(h)$ functions are more irregular than $\psi(h)$ because $\alpha(h)$ is more sensitive to the spatial configuration of the flow lines (Fig. 5.5, 5.6). The $\alpha(h)$ of storm S2 is higher for h values between 80 and 200 m than at the river border (h = 0 m). This feature reflects the fact that the fraction of connected pixels with concentrated flow is larger between 80 and 200 m from the river than at the river border (Fig. 5.4 & 5.6). Such a break in the decline of a connectivity function with distance is not uncommon (Western et al., 2001). In the case of the pattern of S2, the break is partly caused by connected pathways that run more or less parallel to the river (Fig. 5.4). These pathways typically consist of flow on roads and along terrace borders.

The $\alpha(h)$ function of S1 approaches that of random pattern S0, confirming the observation that a dense terrace network can strongly reduce the overland flow to the river (Fig. 5.6). It should be noted however, that the terraces do not prevent overland flow on dirt roads (Fig. 5.1 & 5.4, S1). The flow lines on roads do not influence the $\alpha(h)$ function of S1 because they do not run all the way to the river (Fig. 5.4). The $\psi(h)$ function, however, captures all flow lines, including those on roads, so that the $\psi(h)$ of S1 is larger than that of S0 at $h \geq 10$ m (Fig. 5.5, S0 & S1).

5.3.3 Effects of connectivity on discharge

The presence of terraces leads to a strong reduction in connectivity, contributing area of concentrated flow and peak discharge (Table 5.2). It should be noted that the error of the estimated peak discharge is unknown and may be considerable, especially in the case of scenarios S1 and S2. No field data are available that can be used to test the quality of the estimates. Although scenarios S1 and S2 are based on acceptable assumptions, they do not represent actual field observations of concentrated flow like the data set of E4.

Table 5.2. Production of concentrated flow and discharge to the river (southeastern subcatchment).

Event	A_Q [a] (ha)	A_c [b] (ha)	A_c as % of agricultural area	Runoff production [c] (mm)	Runoff production [c] (m^3)	Q_p [d] ($dm^3\ s^{-1}$)	I_ψ [e] (m)	I_α [e, f] (m)
E1	3.62	0.24	0.2	0.7	1.7	1.7	0.5	1.6
E2	0.03	0.00	0.0	2.0	0.6	0.0	0.0	0.0
E3	7.67	0.84	0.7	5.3	44	9.1	1.1	6.3
E4	15.2	2.01	1.9	13.2	265	20	2.9	8.5
S0	15.2	0.55	0.5	15.6	85	11	1.1	2.6
S1	11.1	0.61	0.5	15.8	96	13	1.8	3.0
S2	17.7	12.2	11.2	13.4	1655	114	4.4	62.1

[a] Area producing concentrated flow.
[b] Contributing area of concentrated flow to the river.
[c] Average depth and total volume of runoff of the area connected to the river (A_c). E2 is an exception: $A_c = 0$ and the reported figures reflect the runoff production of A_Q.
[d] Peak discharge into the river (Eq. 5.5).
[e] Integral connectivity scale. The subscript ψ refers to hydrological connectivity and α to contributing area.
[f] Values based on $h \in [0,400]$. See also the caption of Fig. 5.6.

For a rainfall event in the order of 90 mm the contributing area to the river system (A_c) has increased by a factor 3.2 between 1956 (S1) and 2006 (E4; Table 5.2). If all terraces were removed (scenario S2), the contributing area may further increase by a factor 6.0 compared to 2006 (E4), since more roads and thalwegs become connected to the river (Fig. 5.4 & 5.6). The sharp increase in contributing area from E4 to S2 is due to the different spatial pattern rather than the extent of areas generating concentrated flow (A_Q), which increases from 15.2 to 17.7 ha (+16%). I_ψ increases by as much as 50%, I_α by 630% and the estimated peak discharge by 470% (Table 5.2, Fig. 5.5 & 5.6). Such breakpoints in hydrological response related to spatial patterns, thresholds and scale have been documented previously and reflect the non-linear behaviour of rainfall-runoff processes (Cammeraat, 2004; Bracken & Croke, 2007; Frot et al., 2008).

Although the terraces are quite effective in reducing runoff to the river, the blockage of overland flow is not absolute. Storms like E4 (T = 8.2 years) and larger result in terrace failure and overflow during the event. Field data indicate that widespread terrace failure will occur during events with a return period in the order of 10 years and more (Cammeraat, 2004; Bellin et al., submitted).

The peak discharge into the river is low because the fraction of the agricultural land that becomes connected during the events is small (A_c; Table 5.2). Even during E4 this fraction does not exceed 2%. Despite the size of E4 (92 mm), most of the rain falls at intensities that are below K_{sat} (Fig. 5.7).

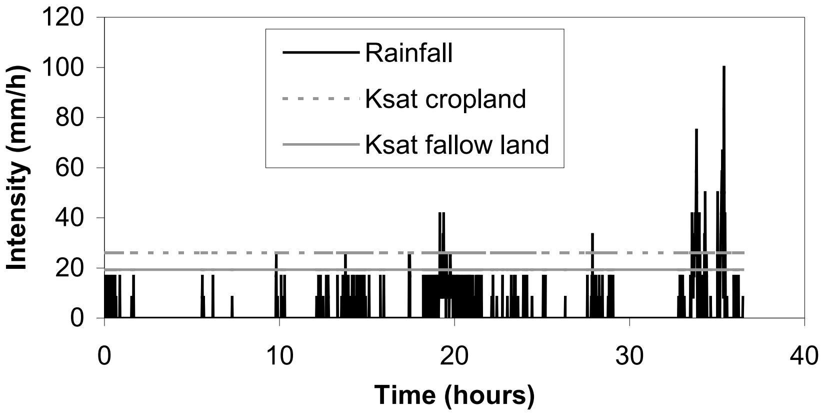

Figure 5.7. The rainfall intensity of storm E4 compared to the average K_{sat} of cropland (26.1 mm h^{-1}) and fallow land (19.3 mm h^{-1}).

The last three peaks produce 21.9 mm of rain with an intensity above the K_{sat} of both fallow and cropland, which is spread over 1.8 hours. This explains why A_c is smaller than might be expected from the large rainfall sum. Only for S2 does A_c reach a somewhat higher value of 11%. A crest stage recorder in the Cárcavo river downstream of the catchment indicated a peak discharge of 1.59 m^3 s^{-1} during E4 (Hooke & Sandercock, 2007). The catchment draining to that point was 5.7 km^2, 4.3 times larger than the southeastern subcatchment (Fig. 5.1), and consisted for about 80% of cropland and for 20% of reforested areas and semi-natural vegetation. If the runoff had been produced homogeneously in space, then the peak discharge of the southeastern subcatchment would have been in the order of 360 dm^3 s^{-1}. The estimated contribution of the agricultural area is an order of magnitude smaller at 20 dm^3 s^{-1} (Table 5.2). This large gap suggests that there is another and more important source of flow that contributes to the catchment discharge. We think that a considerable amount of flow is generated in the river channel itself (Fig. 5.1). Unfortunately, no observations are available to test this hypothesis. In any case, it appears that without terraces, the overland flow from the agricultural area increases an order of magnitude and forms a significant contribution to the peak discharge of the river (S2, Table 5.2).

Although the discharge from the agricultural land is quite low (Table 5.2), this is not uncommon in semi-arid catchments (Rockström & Valentin, 1997; Nasri et al., 2004; Chapter 3). For example Nasri et al. (2004) showed that water harvesting structures can store essentially all runoff within the cropland so that it remains disconnected from the river. Another study from the Cárcavo basin indicates that erosion and runoff on the croplands is actually lower than in reforested and semi-natural areas (Castillo et al., 2002). The authors argue that this remarkable observation is probably explained by the relatively gentle slope of the croplands and their high infiltration rate due to regular tillage and terraces (Castillo et al., 2007). Only if the amounts of high intensity rainfall are large enough will the peak discharge and connectivity to the river become significant.

5.3.4 Concentrated flow in relation to rainfall and landscape properties

A number of overland flow statistics are significantly correlated with storm parameters, such as P, EI_{30} and E_k (Table 5.3). The logarithmic shape of some of them reflects the fact that the fraction of the area producing overland flow and the connectivity functions cannot exceed one (Eq. 5.2 & 5.4).

Table 5.3. Significant relations between rainfall properties and concentrated flow as observed for four storms (southern subcatchment).

Relation	r^2	p	Curve
A_Q [a] vs. EI_{30} [b]	0.99	0.004	logarithmic
L_{50} [c] vs. P [d]	0.96	0.017	logarithmic
L_{max} [e] vs. EI_{30}	0.99	0.002	linear
n [f] vs. EI_{30}	0.93	0.032	logarithmic
I_ψ [g] vs. EI_{30}	0.98	0.008	linear
I_α [h] vs. EI_{30}	0.96	0.017	logarithmic

[a] Area producing concentrated overland flow.
[b] Storm erosivity index.
[c] Median length of the flow lines.
[d] Event precipitation sum.
[e] Maximum distance of overland flow to the river.
[f] Number of flow lines.
[g] Integrated connectivity scale (hydrological connectivity)
[h] Integrated connectivity scale (contributing area)

The concentrated flow is not distributed homogeneously over the different land use classes in the catchment. There is a concentration of flow on compacted surfaces like dirt roads and farmyards, which cover just a few percent of the catchment. These surfaces have the lowest infiltration capacity and surface roughness and are therefore the first locations to produce concentrated flow. The fraction of flow on roads ranges from 27 to 53% and decreases with event magnitude (Table 5.4). Event E2 was excluded from

Table 5.4 because traces of flow on roads were no longer visible at the time of the field survey. The relative importance of runoff on roads has been widely documented (Wemple et al., 1996; MacDonald et al., 2001; Arnáez et al., 2004; Croke et al., 2005). Roads, terraces, field boundaries and tillage often enforce flow patterns that contrast with the overall topography (Fig. 5.1, 5.2, 5.3 & 5.4).

Table 5.4. The area of grid cells producing concentrated flow (A_Q) per land use class and event (southern subcatchment). A_Q is expressed in ha.

Event	Roads	Cropland [a]	Fallow [a]	Sum
E1	2.7 (54%)	1.6 (32%)	0.7 (14%)	5.0 (100%)
E3	4.7 (39%)	5.2 (43%)	2.2 (18%)	12.1 (100%)
E4	6.4 (27%)	13.7 (59%)	3.2 (14%)	23.3 (100%)

[a] The cropland area measures 108 ha and the area of fallow land 44.5 ha.

With increasing storm intensity and duration, the fallow and cropland start producing runoff as well. The area producing concentrated flow increases more rapidly with storm size (EI_{30}) on cropland than on fallow (Table 5.1 & 5.4). The difference is probably due to micro-topography and soil properties. Due to regular tillage, the cropland has a higher infiltration rate and a greater surface roughness (Fleskens & Stroosnijder, 2007; Chapter 3). The infiltration rate on fallow is lower due to crusting. The crust is formed during the first rains after the last tillage pass, and transforms from a structural crust to a thicker and more stable depositional crust if the field is kept fallow long enough (Valentin & Bresson, 1992). The cultivated fields therefore require a higher rainfall intensity and amount to exceed the infiltration rate and surface storage and to produce overland flow. Furthermore, terraces are more likely to be maintained on cropland, further increasing surface storage. According to the observed events, the area fraction with concentrated flow becomes larger on cropland (0.12 for event

E4) than on fallow land (0.071) once the higher cropland thresholds are exceeded (Table 5.4).

Analysis of topographic variables and the origin of flow lines (i.e. rills and gullies) reveals no threshold for concentrated flow in terms of upslope drainage area at the scale of the analysis. Consequently, the data show no slope-area threshold as described for gullies by among others Vandekerckhove et al. (1998). Of the flow lines 33% have an upslope area of 100 m^2 or less (n = 760). This means that the actual threshold of upslope drainage area is quite low and is a function of variability at the plot scale rather then topography at the field scale. The result supports the working hypothesis that was used, i.e. that the actual contributing area of runoff roughly equals the area of 100 m^2 grid cells through which the observed flow lines run.

5.3.5 The use of connectivity measures

The application of connectivity functions adapted from Western et al. (2001) demonstrates that connectivity functions provide a versatile tool to quantify hydrological connectivity and to characterise the rainfall-runoff behaviour of small semi-arid catchments. Our results show that the contributing area of flow can be expressed as a connectivity function, which illustrates the close relationship between hydrological connectivity and contributing area. Furthermore, our results point out that the assumption of catchment-wide runoff and a constant runoff coefficient is not valid in semi-arid areas. Therefore, simple models that are based on a fixed contributing area and a constant runoff coefficient, like the curve number method (Soil Conservation Service, 1985), or that use area weighted infiltration properties produce rather poor predictions, especially for individual events (Mueller et al. 2007, Chapter 3).

Our findings are in line with other publications that demonstrate that rainfall-runoff and erosion models can be improved by incorporating the impact of connectivity (Mueller et al., 2007; Verstraeten et al., 2007). A prerequisite is the availability of spatial data sets, which can be collected by means of field surveys and remote sensing. Examples of the use of satellite imagery to incorporate the effect of vegetation patterns are provided by Mueller et al. (2007) and Lesschen et al., submitted). Patterns of soil moisture and concentrated flow could be derived by remote sensing as well in areas with low vegetation cover.

A more empirical approach to improve discharge estimates would be to establish the $\alpha(h)$ function of a catchment from field surveys of overland flow. Our results suggest that the link between EI_{30}, the integral connectivity scale (I_{α}) and $\alpha(h)$ can be used to estimate the contributing area of concentrated flow to the river or catchment outlet from rainfall data (Table 5.3 & Fig. 5.6). The $\alpha(h)$ function effectively summarises the extent of the contributing area as well as its distribution in terms of distance towards the river. This information of distance can be translated to an estimate of runoff travel time and time to concentration. It will probably be more accurate for the estimation of travel times to define h not as the Euclidian distance to the river, as was done here (Eq. 5.4), but as the flow distance.

Once the properties of the contributing area are determined, the next step is to estimate the runoff production at the pixel scale and to apply a technique to deal with the transmission of the runoff to the river, taking into account the rainfall variability of the event and runoff travel times. This could be an approach similar to the CRUM model (Reaney et al., 2007).

5.4 Conclusion

It was shown that the degree of connectivity in spatial patterns of concentrated overland flow can be quantified by means of connectivity functions. The hydrological connectivity function reflects the presence of connected features, whereas the contributing area function characterises the contributing area to a specific object like a river system or catchment outlet. The integral connectivity scale, which summarises the connectivity function in a single number, is related to storm magnitude. It is possible to convert storm magnitude into an estimate of contributing area and travel distance distribution to the catchment outlet. Hence, connectivity functions can be used to develop and improve rainfall-runoff models.

Terrace failure results in a strong increase in hydrological connectivity and discharge in the agricultural catchment studied. The lack of terrace maintenance between 1956 and 2006 has increased the contributing area to the river system by a factor 3 for an event with a return period of 8 years. A further increase by a factor 6 compared to 2006 is expected in the case that all terraces fail and become permeable to overland flow. The discharge from the agricultural land into the river is small during small and moderate storms due to the limited contributing area. Widespread overflow and terrace failure occurs only for storms with a return period in the order of 10 years and more. Without terraces, the overland flow to the river would increase an order of magnitude and would form a significant contribution to the peak discharge of the river.

A large part (27-54%) of the concentrated flow is produced on dirt roads. The fraction of flow that occurs on roads decreases with increasing storm magnitude, whereas it remains stable on fallow land and increases sharply on croplands. There are no distinct topographic thresholds for the production of concentrated flow at the scale of the digital elevation model (>=100 m^2).

CHAPTER 6
THE IMPACT OF COVER CROPS ON THE ORCHARD WATER BALANCE

6.1 Introduction

The soil of orchards in temperate climates is commonly covered by grass or other vegetation. This is not the case in semi-arid regions, where farmers fear the competition between cover crops and the tree crop for the scarce water resources. Hence, they generally prefer to keep the orchard floor bare, which leads to accelerated soil erosion on sloping land. Cover crops in orchards can either increase or decrease the individual terms of the water balance compared to conventional practices (Ingels et al., 1998). The magnitude of these changes in the individual terms depends on climate, cropping system and soil conditions. The overall impact on the water uptake by the main crop is therefore site-specific. The most frequently cited benefits of cover crops are an increase in infiltration and hence a decrease of runoff on sloping land as well as an improved soil structure (Ingels et al., 1998; Hartwig & Ammon, 2002; Pastor, 2004). The most notable drawback is the potential increase in water use through transpiration of the cover crop, which may be higher than the evaporation from a bare soil and can therefore increase the water stress of the tree crop during dry spells (Ingels et al., 1998; Hernández et al., 2005). The implementation of a cover crop can be considered impracticable if this leads to a reduction in the water uptake by the main crop (i.e. water stress), yield or profit margin.

Still little is known about the threshold conditions for the successful implementation of cover crops in orchards, both in physical and economical terms, as discussed in §2.4.3 and §2.4.4). Publications that discuss the use of cover crops mention that competition for water may be a serious issue in rainfed systems (Ingels et al., 1998, Hartwig & Ammon, 2002). However, these sources hardly provide field evidence about where and when competition is a problem and leads to a significant yield decline. Hartwig & Ammon (2002) state that competition for moisture is minimal in orchards in areas with more than 1100 mm of annual rainfall, or when the cover crop is killed early in the growing season. A field study from central Spain with an annual rainfall of 490 mm seems to support this finding, as three cover crop treatments did scarcely compete for water and olive yields were reduced about 10% compared to conventional tillage during the five years of the experiment (Hernández et al., 2005). The cover crop was killed off or mowed back to a few centimetres at the end of spring. Cover crops are probably unfeasible at an annual rainfall of 300 mm or less, as was illustrated by the water balance study and field observations presented in Chapter 3.

Crop yield data from Jaén, Spain, show that mowing the orchard floor vegetation in spring may decrease olive yields compared to regular tillage (2-4 passes per year, Pastor et al., 2004). Chemical weeding, however, was found to increase yields. More field evidence is required to delineate the physical thresholds for the application of cover crops, and to find out which management strategies are most effective to minimise the competition for water as well as adverse yield effects.

It can be noted that unlike for annual cropping systems, no widely tested water balance and crop growth models are available for tree crops. An orchard does not have a uniform canopy cover, but a heterogeneous pattern made up by the tree canopy and the orchard floor. The fraction of canopy cover is site-specific, and the properties of both the canopy and soil cover change throughout the year, depending on crop development and orchard floor management (Connell et al., 1996; Goldhamer, 1996; Tubeileh et al., 2004). One of the challenges for modelling the water balance of this complex system is the calculation of evapotranspiration, as was illustrated in §2.2.4 under the heading "Olive, almond and grapevine". Another is the partitioning of the water uptake and evapotranspiration between the trees and a living cover crop (Tjhie, 1991; Annandale et al., 2003).

The aim of this chapter is to compare the water balance of a treatment consisting of regular tillage of the traffic lane and a treatment with a cover crop. The specific objectives are (1) to compare soil moisture levels between a tillage and cover crop treatment; (2) to put the observed differences into perspective by simulation of the water balance of the traffic lane and (3) to study the effect of the length of the cover crop growing season on soil moisture levels during the summer. The hypothesis is tested that the length of the cover crop growing season influences the duration and degree of water stress of the tree crop during the summer period. The study is based on a field experiment in an olive (*Olea europea* L.) orchard near Sevilla, south-east Spain.

6.2 Materials and methods

6.2.1 Field experiment

The olive orchard is located close to Benacazón in the province of Sevilla, Spain (37°20' N; 6°13' W). The average annual temperature is 17.9 °C, the annual precipitation 520 mm and the potential evapotranspiration 1220 mm y^{-1} (data derived from the ATEAM climate database: Mitchell et al., 2002). The soil is classified as a calcaric regosol (FAO, 1990) and has a loamy texture. The tree spacing is 6.5 by 5 meters and the plantation age is approximately 20 years. According to the manager of the field experiments, the orchard is under drip-irrigation from spring to September (personal communication with J.A. Gómez).

The soil of the tree line was kept weed-free by herbicides and was not ploughed. Two treatments were applied in the traffic lane, which is a strip of effectively four meters wide that is not covered by the tree canopy (Fig. 6.1).

Figure 6.1. Overview of the tillage treatment (upper photo) and cover crop treatment (lower photo) in January 2007.

The lanes of the treatment run in the slope direction. They have a convex profile curvature and a straight plan curvature. The tillage treatment (TI) consisted of regular tillage over a depth of 10-15 cm (2-4 passes per year), in order to suppress weeds in the traffic lane (Fig. 6.1). A duckfoot chisel was used, so that the soil was not turned upside down, but was merely lifted and loosened, cutting the roots of the weeds. The cover crop treatment (CC) consisted of ryegrass (*Lolium rigidum* Gaudin), which was sown at the start of the winter in 2004, 2005 and 2006. Chemical weeding took place on 23-4-07 and 18-3-08. The remnants of the cover crop were left as a mulch and the soil was not tilled during the entire period. Yields were measured for 20 trees per treatment in October 2007.

Soil moisture was measured at an upslope and midslope position for both treatments using theta-probes (model ML2x, Delta-T Devices Ltd., Cambridge). The slope was about 50 m long with a gradient of 7% at the upslope position and 17% at the midslope position. The probes were inserted sub-horizontally (the prongs pointing upward in order to prevent disturbance of the signal by water flow along the cable) at a depth of 45 cm. This allowed studying the percolation of rainfall to the subsoil, which forms the main water source for the trees during dry spells (see also Chapter 3). Furthermore, the depth of 45 cm was chosen to avoid damage to the sensors and cables in the lane with regular tillage. The sensor readings were stored every hour on a data logger. Meteorological variables were recorded by an automated station at the foot of the slope in the traffic lane. The measurements included air temperature, relative humidity, wind speed and incoming solar radiation.

6.2.2 Simulation of the water balance

The water balance of the traffic lane was simulated based on the observed weather and soil properties in 2007 (from day of the year (*DOY*) 19 to 242).

The model is limited to the traffic lane where we consider only the water uptake by the cover crops. The difference in water availability for the tree crops between the two treatments will then be discussed based on the differences in evapotranspiration and drainage between the cover crop and tillage treatment.

The simulation exercise was done with the BUDGET water balance model (Raes et al., 2006). The model has been shown to be robust and is able to simulate the water balance of a bare soil or a single crop. BUDGET has been successfully applied in a range of environments (Raes et al., 2006; Meerkerk et al., accepted). The lysimeter experiment presented in Chapter 3 showed that BUDGET is capable to simulate the water balance of the bare soil in a realistic way, even if some of the soil parameters, which cannot be derived from field measurements, are left uncalibrated. The water balance of this application can be expressed as (Eq. 6.1):

$$\Delta S = P - Q_{\text{off}} - D - ET_{\text{a}} \qquad (6.1)$$

Where ΔS is the change in soil moisture storage over a given time interval, P is precipitation, Q_{off} is runoff, D is drainage, and ET_{a} is the actual evapotranspiration (Allen et al., 1998). All terms are expressed in mm. A more detailed description of BUDGET is provided in Chapter 3, Raes (2002) and Raes et al. (2006). The infiltration of run-on was ignored because it could not be quantified or simulated given the available data. This is a reasonable assumption, since mapping of rills and sediment deposition in the Cárcavo basin (Chapter 3) showed that run-on is only important upslope of dams or terraces .

The TI treatment was represented in the model as a bare soil. The soil properties of the test lanes were measured in 2005 for the 0-10 cm and the 10-20 cm layers (Table 6.1). The measurements included texture, organic matter, saturated hydraulic conductivity and water retention curves. The same soil input was used for both treatments, so that the TI and CC runs are identical, except for vegetation cover. The soil was represented in the model by 9 layers, with a thickness (from top to bottom) of 5, 5, 5, 5, 10, 10, 10, 25 and 25 cm. The deeper soil layers were assumed to have the same properties as the 10-20 cm layer. Runoff was estimated with the curve number method, which is incorporated in the BUDGET model (Soil Conservation Service, 1985). A curve number of 65 was applied for both treatments.

Table 6.1. Soil properties of the orchard used for the model runs.

Depth (cm)	Sand ($g\ kg^{-1}$)	Silt ($g\ kg^{-1}$)	Clay ($g\ kg^{-1}$)	OM [a] ($g\ kg^{-1}$)	WP [b] ($m^3\ m^{-3}$)	FC [c] ($m^3\ m^{-3}$)	SAT [d] ($m^3\ m^{-3}$)	K_{sat} [e] ($mm\ h^{-1}$)
0-10	506	340	154	13	0.11	0.28	0.39	11
10-20	363	436	201	8	0.08	0.26	0.37	39

[a] Organic matter
[b] Soil moisture at wilting point ($pF = 4.2$)
[c] Soil moisture at field capacity ($pF = 2.0$)
[d] Soil moisture at saturation ($pF = 0$)
[e] Saturated hydraulic conductivity

The evapotranspiration from the vegetated surface is modelled in BUDGET using the FAO approach, based on the so-called reference evapotranspiration (ET_0) and crop factors (K_c; Allen et al., 1998; Raes et al., 2006). See also §2.2.2: "Crop water requirements". The ryegrass cover crop has a K_c value of approximately one, which corresponds to an albedo of 0.23 (Allen et al., 1998). The potential crop evapotranspiration (ET_c) and the actual crop evapotranspiration (ET_a) are derived from ET_0. ET_c is the product of K_c and ET_0. ET_a becomes smaller than ET_c in times of water shortage.

It was assumed that the ryegrass albedo is constant throughout the year and applies both to the growing cover and to the mulch after chemical weeding. The albedo of the bare soil was estimated at 0.20 when dry and 0.11 when wet. These figures were selected based on albedo values of comparable soils combined with photos from the orchard (Chapter 3; van Kraalingen & Stol, 1997; Post et al., 2000).

Due to gaps in the weather data, the simulations were restricted to the following days in 2007: day of year (*DOY*) 19-242, excluding 95-113, 127-129, 134-137, 148-157, 180-200. For gaps of just one day, the missing weather data was interpolated from the preceding and next day. This concerns *DOY* 120, 132, 141, 162 and 169. The larger gaps were omitted from the run. For example, the model run assumes that day 201 follows directly after day 179, given the gap of day 180-200. The implications for the interpretation of the model results are discussed in §6.3.3.

Three runs were distinguished that differ with respect to the date on which the cover crop is killed. These dates are the end of March (CC-1), the end of April (CC-2) and the end of May (CC-3).

Additional simulations were done to examine the sensitivity of the model to rooting depth and evaporation depth. Both were set at either 30 or 40 cm and a number of different combinations were considered. The mulch cover outside the growing season was set at 35%. BUDGET calculates the effect of a mulch on soil evaporation as follows (Eq. 6.2, adapted from Allen et al. 1998):

$$E_0 = (1 - m / 2)\ ET_{\text{bare soil}} \qquad (6.2)$$

Where E_0 is the potential evaporation, m is the fraction of the soil covered by the mulch, and $ET_{\text{bare soil}}$ is the potential evapotranspiration of the bare soil.

6.3 Results and discussion

6.3.1 Field observations

The amounts of rainfall in 2006 and 2007 around Sevilla were within the range of one standard deviation of annual precipitation. The annual sum amounted to 670 mm in 2006 (average of 6 stations), which is 0.98 standard deviations above the long term average (1971-2000, ATEAM database, New et al., 1999). Rainfall during January - October 2007 was 370 mm, which is 0.87 standard deviation below the long term annual average.

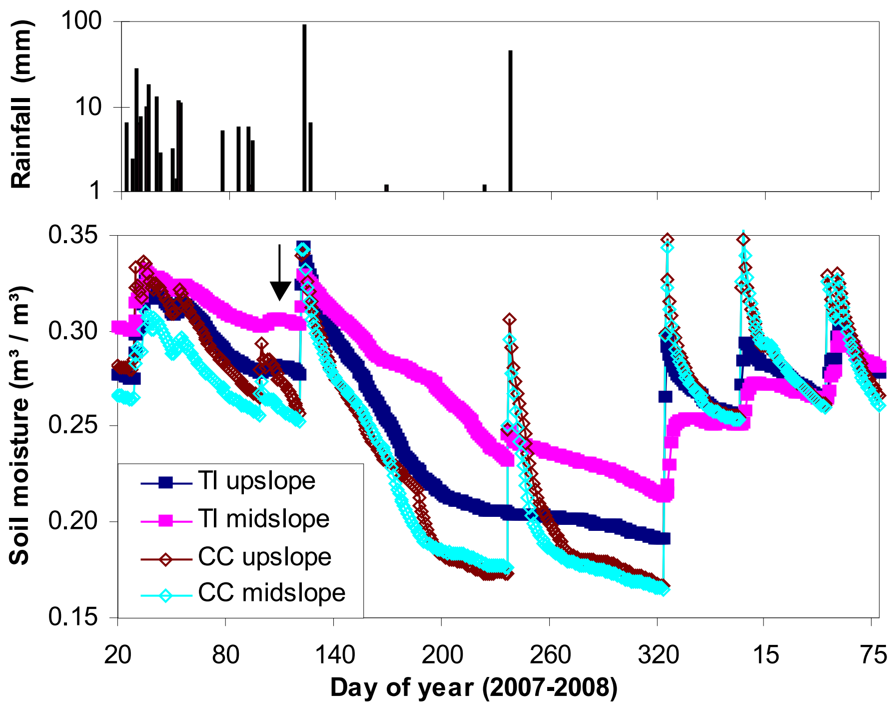

Figure 6.2. Daily rainfall and soil moisture at 45 cm depth in the traffic lane. TI = tillage treatment; CC = cover crop treatment. The arrow indicates the date on which the cover crop was killed. Note that the rainfall record is restricted to days 19-242 in 2007 and is not complete (see also §6.2.2).

The soil moisture content is at field capacity during the winter season (*DOY* 325 - 140; Fig. 6.2, Table 6.1). The rainfall during the period October 2006 - January 2007 was 380 mm.

The CC treatment shows systematically lower soil moisture levels than the TI treatment, except for the winter period of 2007-08 (November - February). The midslope sensor of the tillage treatment returns markedly higher readings than the rest up to November 2007. The difference compared to the cover crop reaches a maximum of 0.085 m^3 m^{-3} on day 200. The CC treatment shows a greater oscillation of soil moisture in time (Fig. 6.2). The most likely explanation is that the total amount of water extracted by plants is larger for the CC treatment and hence, the decrease in moisture following recharge is larger than for the TI treatment. This difference can only be caused by evaporation or transpiration. At a depth of 45 cm it is unlikely that the decrease in soil moisture is due to evaporation. This is for example illustrated by the lysimeter experiment discussed in §3.3.2.

It appears that the trees of the CC treatment extract more water from the subsoil during the dry season than those of the TI treatment. The difference between subsequent highs and lows in soil moisture is considerably larger for the CC treatment (*DOY* 113-320; Fig. 6.2). The cover crop is dead during this period and does not extract any water, so that the difference can only be due to a larger water uptake by the trees of the CC treatment. A more detailed inspection of the data reveals that the increase of soil moisture after rainfall is systematically greater for the CC treatment (Table 6.2). It therefore appears that the cover crop and subsequent mulch increase the percolation of rain water into the sub soil. This could be due to increased infiltration compared to the TI treatment. Nevertheless, no final conclusions can be drawn due to the low amount of observations and the limited time of measurement.

Table 6.2. Average increase in soil moisture at 45 cm due to rainfall.

	ΔS (m^3 m^{-3})	
Period (*DOY*)	Tillage	Cover crop
28-35	0.042	0.047
38-40	0.000	0.004
51-54	0.004	0.010
121-123	0.047	0.087
236-238	0.007	0.126
323-330	0.063	0.180
1-7	0.028	0.097
48-56	0.036	0.068

The difference in the soil moisture response to rainfall increases over the course of 2007. Most notable is the large rainfall on day 237, for which the recharge is minimal for TI and outspoken for CC (Fig. 6.2). Apparently, the rainfall did not penetrate up to the depth of the upslope TI sensor. The reason may be that the TI treatment generated more runoff and less infiltration than the CC treatment. The beneficial effect of cover crops on soil structure and porosity is widely documented (Ingels et al., 1998; Hartwig & Ammon, 2002). A field visit in October 2006 showed that the CC soil had a loose, fine structure, whereas the plough layer of TI was massive and easily compacted by traffic when moist.

The results show that more soil moisture was taken up from the subsoil by plants in the CC treatment, as illustrated by the difference between the maximum and minimum soil moisture content of the treatments, which is on average 0.18 m^3 m^{-3} for CC and 0.13 m^3 m^{-3} for TI (Fig. 6.2). It therefore appears that the CC treatment increases the water uptake at 45 cm depth compared to TI.

Although the soil moisture is lower for the CC treatment during the cover crop growing season in 2007 (day 19-113), the observed values are well above levels associated with plant water stress. The soil moisture did not drop below 0.25 $m^3\ m^{-3}$ (pF = 2.1). The results therefore suggest that the cover crop does not limit the water uptake of the trees from the subsoil during the winter.

The olive yield in October 2007 was on average 76.5 kg $tree^{-1}$ for the trees bordering the CC lane and 83.6 kg $tree^{-1}$ for those bordering the TI lane. The difference is statistically significant at $p = 0.05$. Nevertheless, the difference is not very pronounced and olive yields are highly variable between years. Therefore no conclusion can be drawn on the yield effect.

There are many other factors that could have caused the observed yield difference. For example, Ingels et al. (1998) mention that winter annual grasses require large quantities of nitrogen, which can delay the nitrogen availability to the main crop and reduce its vigour. Hence, the overall effect of the cover crop on tree vigour encompasses more than just water availability. In any case the observed yield decrease under the cover crop treatment is small and might be acceptable when erosion control has a high priority or when it is offset by an increase in profit margin compared to regular tillage. Given the alternate bearing property of olive crops and the high interannual rainfall variability, a longer series of yield data is required to verify if the observed decline is systematic.

6.3.2 Water balance simulations

The simulation results are discussed here within the framework of the model of the traffic lane, which excludes the water uptake by the trees. The simulation period is restricted to *DOY* 19-242 in 2007. The simulations that were done to investigate the sensitivity of the model to evaporation depth and cover crop rooting depth are not discussed separately, because the model output is similar. All results presented here are based on an evaporation depth and cover crop rooting zone of 40 cm unless indicated otherwise.

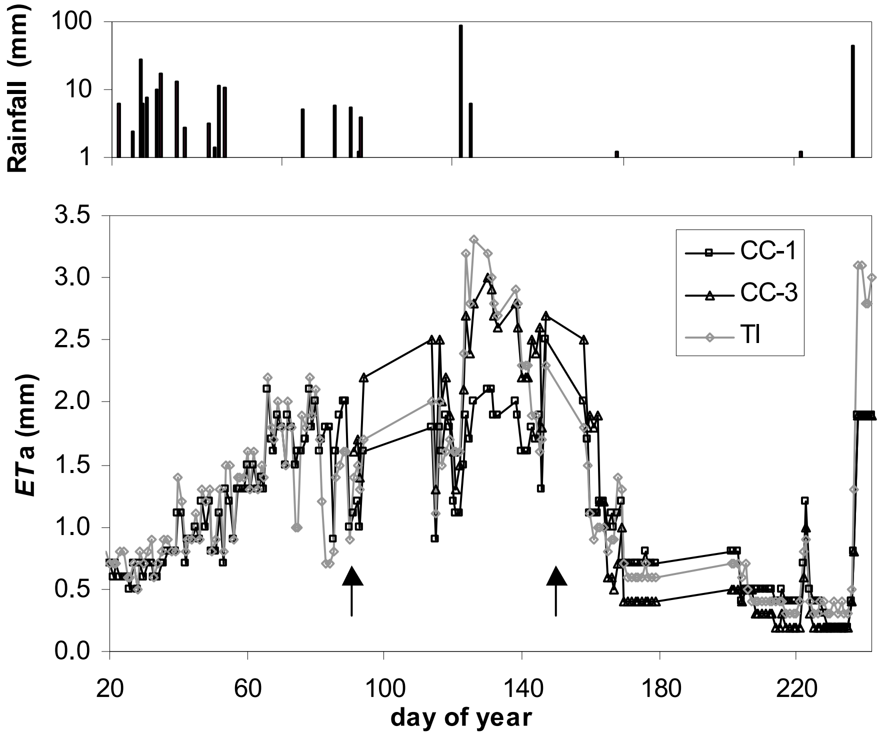

Figure 6.3. ET_a of the tillage and cover crops treatments. The cover crop is killed on *DOY* 90 (CC-1) or 151 (CC-3), as indicated by the arrows. CC-3 is only plotted from the first arrow onwards. Prior to that point, the curve is the same as of CC-1.

The simulations reveal a number of differences between the treatments (Fig. 6.3). The ET_a of the bare soil (TI) is on average 8.3% larger than that of the cover crop under wet conditions (*DOY* 20-73). This is directly related to the lower albedo of the soil surface: more solar energy is absorbed so that ET_c and ET_a are larger. Because the bare soil dries out more quickly, ET_a becomes smaller than that of the CC treatment at the start of dry spells (i.e. day 74, 84, 161). The reason is that the CC soil has more moisture left to evaporate at that point. Hence the vegetation cover moderates ET_a and delays the date at which the water in the top soil is depleted during droughts. An additional consequence is that the vegetation cover results in lower soil temperatures compared to an orchard floor consisting of bare soil (Snyder & Connell, 1996; Ingels et al., 1998).

The mulch of the residues of the cover crop reduces the ET_a of the CC treatments during the summer (Fig. 6.3). CC-3 has the longest cover cop growing season so that cumulative ET_a is highest and drainage lowest (Table 6.3).

Table 6.3. Sum of drainage below 40 cm (D_{40}) and ET_a for 167 days from the period *DOY* 19-242. Due to missing data, 57 days are excluded (see §6.2.2).

Run*	ET_a (mm)	D_{40} (mm)
TI	203	106
CC-1	185	111
CC-2	190	106
CC-3	196	105

* CC-1: cover crop killed at the end of March; CC-2: at the end of April; CC-3: at the end of May.

The effect of the mulch of the cover crop treatments on ET_a during the summer is larger than the effect of the difference in albedo, as illustrated by *DOY* 124-142 and *DOY* 238-242 (Fig. 6.3). During these periods the ET_a of

CC-1 is about 1 mm day^{-1} smaller than that of TI, whereas the decrease in ET_a due to albedo is on average 0.08 mm day^{-1} (*DOY* 19-73). It can be noted that the albedo is a function of soil and crop properties, as well as the humidity of their surface, and varies in space and time. The soil albedo typically ranges from 0.11 to 0.40 when dry and 0.04 to 0.16 when wet (Post et al., 2000). A green vegetation cover has an albedo in the order of 0.12 to 0.23. Therefore, in some cases the ET_0 of the bare soil will be larger than of the cover crop and in other cases it will be smaller. The albedo of the bare soil of the simulations was always lower than that of the cover crop (§6.2.2; Fig. 6.3).

The effect of a mulch consisting of dead grass is more straightforward than the albedo effect. A mulch decreases soil evaporation compared to a bare soil. There are many studies that show the effectiveness of mulches to reduce evaporation (Li, 2003; Mellouli et al., 2000). Although it is clear that the mulch reduces ET_0 and ET_a, the magnitude of the effect cannot be directly evaluated from the simulation. The model result is based on the assumption that the mulch covers 35% of the soil. This translates to a reduction in E_0 of 18% (Eq. 6.2). This is a conservative figure when compared to the field situation in Oct 2006, when the mulch covered close to 100% of the soil.

Once the cover crop of CC-3 is killed on day 151, ET_a and soil moisture converge to those of CC-1 (Fig. 6.3 & 6.4). The soil moisture of the CC treatments is similar to the TI treatment during the growing season of the cover crop (Fig. 6.4). When the cover crop of CC-1 is killed, the soil moisture increases by 0.02 to 0.04 m^3 m^{-3} compared to the TI treatment due to the reduction in evaporation losses induced by the mulch. Hence, the results suggest that the water availability is larger for the CC treatment in the summer if the crop is killed on day 90 (CC-1). Treatment CC-3, however, has a similar soil moisture content as the TI treatment during the summer. This is probably because the cover crop is killed during the dry spell (day 126-236): the mulch makes a smaller difference when soil moisture and evaporation rates are low.

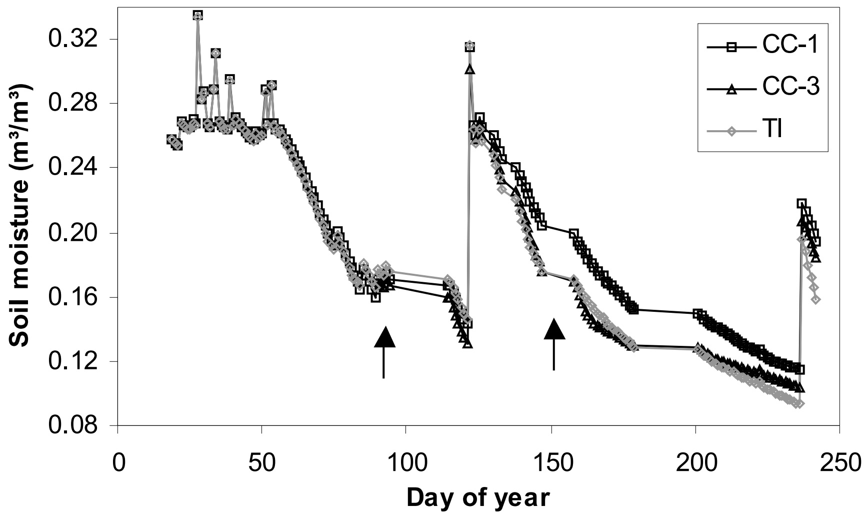

Figure 6.4. The soil moisture of the 0-40 cm layer for tillage and cover crop treatments. The cover crop is killed on day 90 (CC-1) or day 151 (CC-3) as indicated by the arrows.

6.3.3 Comparison of model output and field data

The model cannot be validated in terms of soil moisture because it excludes the water uptake by the trees. Nevertheless, it is possible to compare the modelled soil moisture of the 30-40 cm layer with the measured soil moisture at 45 cm. Given the simplifications of the model, it is not surprising that the agreement between measured and modelled soil moisture is not very strong (Fig. 6.5).

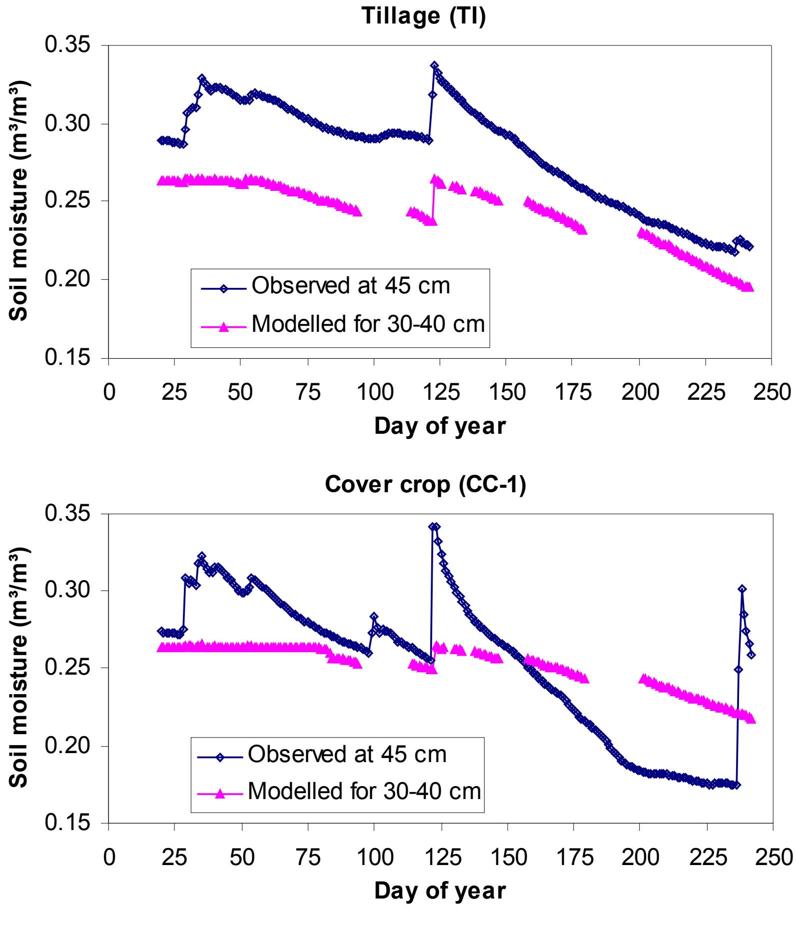

Figure 6.5. Comparison of modelled and observed soil moisture. The field observations represent the average of two measurements per treatment (Fig. 6.2).

Apart from the exclusion of the water uptake by the trees, the disagreement between the model and the observations is probably related to the simplified representation of the soil and the complex processes of infiltration, water extraction, percolation and redistribution. The modelling exercise illustrates that the accurate simulation of the water balance of relatively complex cropping systems such as orchards remains a challenge. It can be noted that the results presented were obtained without model calibration. A calibration exercise would improve the agreement, but not the capability of the model to simulate the soil moisture for locations with different soils and cover crops.

Some of the deviation of the model results compared to reality is caused by the gaps in the weather data. The soil moisture observations show that the difference in moisture before and after the gap is below 2% for all gaps, except for *DOY* 180-200, when it was 3.2% for the CC treatment (Fig. 6.5). Without the gaps, the modelled moisture would have shown a stronger decline after *DOY* 125, and would have better followed the behaviour of the observations.

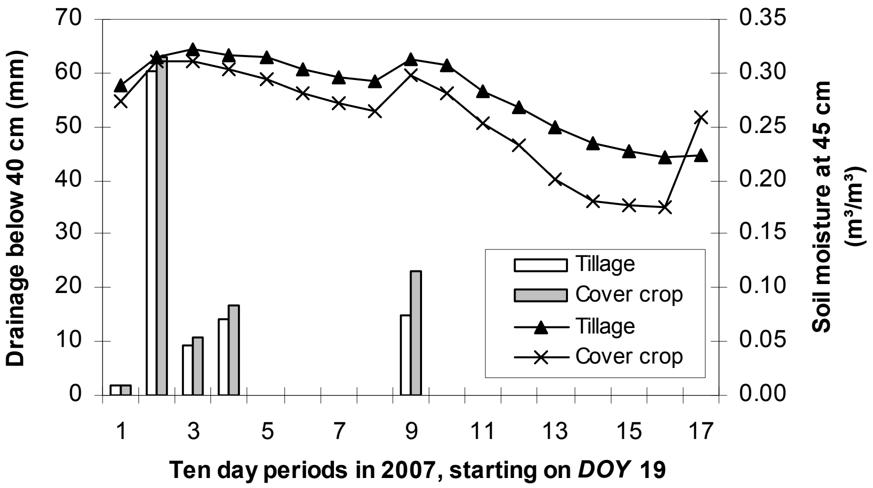

Figure 6.6. Modelled drainage below 40 cm (bars) and observed soil moisture at 45 cm (lines). The model run assumes an evaporation depth of 40 cm and a cover crop rooting zone of 30 cm. The cover crop is killed on day 90 (ten day period 8 in the graph).

The observed soil moisture can also be compared to the simulated drainage from the 0-40 cm layer (Fig. 6.6). As discussed previously, the observed increase in soil moisture at 45 cm following rainfall is systematically larger for the CC treatment (Table 6.2). This means that the simulated drainage below 40 cm should be larger as well. It appears that up to the ninth decade, the model realistically mimics an increase in drainage for the CC treatment (Fig. 6.6). This results from the fact that the simulated ET_a of the cover crop is lower and the soil moisture higher (Fig. 6.3 & 6.4). The large event of day 237 (decade 17 in Fig. 6.6) however caused no drainage below 40 cm according to the model. This is realistic for TI, where the increase in soil moisture at 45 cm was 0.007 m^3 m^{-3}, but not for CC where it was 0.126 m^3 m^{-3} (Table 6.2). It could be attempted to improve the simulated drainage by estimating the runoff of the TI and CC treatment separately.

6.4 Conclusion

The water balance of the traffic lane of an olive orchard was compared for a treatment with regular tillage and one where a cover crop was grown during the winter months. The average annual rainfall of the orchard near Benacazón, Sevilla, is 520 mm. Soil moisture observations at 45 cm depth strongly suggest that the cover crop treatment increases percolation as well as the moisture uptake from the subsoil in the traffic lane. There were no signs of competition for water between the trees and the cover crop during the winter period, while soil moisture levels remained close to field capacity (> 0.25 m^3 m^{-3}). The results suggest that the cover crop treatment increased the water uptake from the subsoil during the summer. The olive yield was on average 8.5% lower compared to the tillage treatment. Additional field observations and a longer measurement period are required to confirm and explain the yield effect. While the yield decline is small, the application of cover crops may be a feasible option. An economic analysis is required to determine the difference in profit margin between the two systems.

Simulation of the water balance reveals two factors that contribute to an increase in available water to the trees under the cover crop treatment. Firstly, the mulch provided by the residue of the cover crop reduces soil evaporation during the summer. Secondly, the cover crop system has a higher albedo and lower ET_c than the bare soil. It should be noted however, that albedo values vary in time so that the effect on the difference in ET_c between the tillage and cover crop treatment is not constant. On locations with a very light coloured soil, a cover crop may actually increase ET_c compared to the tillage treatment. For the orchard at Benacazón, the effect of the mulch on the reduction in ET_a is larger than that of the albedo difference.

The BUDGET water balance model is able to reproduce two important aspects of the water balance of the traffic lane, which are the ET_a and the drainage to the subsoil, which is larger for the cover crop treatment. The simulations suggest that the cover crop treatment will only increase the available water during the first part of the summer when it is killed early in the spring. Otherwise, the mulch is established too late to reduce the ET_a of the traffic lane following the last spring rains.

CHAPTER 7
CONCLUSION

7.1 Main findings

Rainfed orchards cover a considerable part of the agricultural land in the Mediterranean. The sustainability of the cropping system is questionable in dry areas due to a decrease of water availability as a result of global warming and high erosion rates. It is expected that rainfall in the Mediterranean will decrease and become more variable, and that temperatures will increase (Olesen & Bindi, 2002). The modern cropping system is characterised by a decline in SWC structures and an increase in field size (Beaufoy, 2003; Grove & Rackham, 2003).

There are two main strategies to mitigate the degradation and abandonment of orchards in drylands. The first is to increase the water use efficiency of the cropping system and the second is to minimise soil erosion. The results of Chapter 3 show that the evaporation from the top soil forms the largest loss of water in rainfed orchards. It appears that the practice of clean sweeping (i.e. frequent shallow tillage) limits the water availability in orchards on loamy soils with an annual rainfall in the order of 300 mm. Clean sweeping constrains the root growth in the plough layer (upper 15 cm), so that the trees cannot access the water from light rainfall. This is a serious drawback, because a considerable part of the rain events are in the order of 10 mm or less. The percolation of water to the subsoil is more pronounced on stony soils than on marls, due to the low water holding capacity and high stone content.

Runoff is of minor importance for the water availability in dry areas, since the annual runoff coefficient of the orchards is low (0.7 - 5.2%; Chapter 3). Nevertheless, infiltration of overland flow in depressions, and on downslope terraces, can significantly increase the water availability and tree vigour at these locations, illustrating the use of water harvesting techniques.

The vulnerability of rainfed orchards to land degradation and global warming depends on spatial patterns of water availability at different scales. Chapter 4 points out a pattern related to lithology at the regional scale (Murcia province, south-east Spain). It appears that there is a significant concentration of almond orchards on shales and schists and to a lesser extent on marls. The preference for stony soils is probably related to the relatively high water availability, which results from the lower evaporation loss compared to marl soils. The stones promote percolation to the subsoil because they decrease the volume of fine earth and the water holding capacity of the soil. The concentration of orchards on marls is probably due to the mouldability of the soft bedrock, which facilitates the site preparation and planting of new orchards. The relative absence of orchards in limestone and dolomite areas may be explained by the fact that these rocks are not easily mouldable and often coincide with mountainous terrain. Furthermore, soils on limestone and dolomite have a low water retention capacity.

At the scale of the Mediterranean region, patterns of water availability are imposed by the spatial variability in climate. The results of Chapter 4 demonstrate that the canopy cover and tree density of olive plantations are significantly related to a climate indicator such as the humidity index (*HI*, the ratio between annual precipitation and reference evapotranspiration). The canopy cover increases with *HI* and reaches a maximum around $HI = 0.6$. It seems that there is a surplus of water at *HI* > 0.6, which will enable the growth of cover crops without compromising the water availability to the trees. Below this *HI* threshold, the growth of cover crops must probably be restricted to the winter season and to a fraction of the orchard floor, in order to avoid competition with the main crop and significant yield declines.

The decline in traditional SWC structures was studied in an agricultural subcatchment of the Cárcavo basin in Murcia province, south-east Spain (Chapter 5). Connectivity functions were defined and used to investigate the effect of terrace removal and failure on hydrological connectivity and peak discharge. The results show that there has been a strong increase in hydrological connectivity and discharge between 1956 and 2006. During this period, the contributing area of concentrated flow to the river system increased by a factor 3 for an event with a return period of 8 years. A further increase by a factor 6 compared to 2006 is expected in case all terraces become permeable to overland flow. The discharge from the agricultural land into the river is small during small and moderate events due to the limited contributing area. Widespread overflow and failure of intact terraces is limited to events with a return period of 10 years and more. Furthermore, it appears that a large part of the concentrated flow (25-50%) is generated on dirt roads. This fraction decreases with increasing storm magnitude, when a larger part of the croplands become connected.

Chapter 6 examined the effect of the implementation of a cover crop in an olive orchard near Sevilla, Spain. The orchard was irrigated (drip irrigation) and received an annual rainfall in the order of 520 mm. Soil moisture measurements in the traffic lane, at a depth of 45 cm, strongly suggest that the cover crop treatment increased the percolation of water to the subsoil compared to regular tillage. There were no signs of competition for water between the tree crop and cover crop, while soil moisture levels remained close to field capacity during the winter season. Nevertheless, the olive yield was 8.5% lower for the cover crop treatment.

Water balance simulations reveal two factors that can reduce the evapotranspiration from a cover crop compared to the bare soil. Firstly, the mulch of a dead cover crop decreases evaporation after the cover crop season. Secondly, the cover crop has a larger albedo and lower evapotranspiration rate under well watered conditions than the bare soil. The mulch effect is considerably larger than the albedo effect. The magnitude of the albedo effect is site-specific and may be negative if the soil albedo is larger than that of the cover crop.

7.2 Suggestions for further research

The results of the thesis reveal a number of gaps in our knowledge of rainfed perennial crops. In terms of the water balance and water use efficiency in dry areas (300 mm of annual rainfall), the use of no-tillage and minimum tillage in stead of clean sweeping deserves more attention (Chapter 3). The body of literature on this topic is quite restricted, while most of the literature does not deal with perennial crops in marginal areas with low rainfall and no irrigation. Nevertheless, many people depend on the productivity of these marginal areas by means of subsistence agriculture, for example in countries like Syria and Tunisia. A comparative field study of at least a few years would enable to detect the effect of the mentioned tillage techniques on the water balance and yields. Use could be made of sap flow gauges to precisely quantify the transpiration of the trees (Giorio & Giorio, 2003).

Our knowledge on the spatial variability in plant available water and orchard properties could be improved by expanding the data sets of Chapter 4. At the regional scale, it would be useful to study the orchard water balance on a larger range of soil types. At present, we have data for marls and stony soils on slate, which are however limited to a few locations. It would be interesting to develop a map indicating the spatial variability in water availability related to climate at the Mediterranean scale. Coupled with a conservative climatic threshold for cover crops, this would provide a first impression of locations where cover crops may be successfully incorporated in rainfed orchards. At the same time, more field experiments are required to verify the climatic threshold for cover crops in olive orchards, as identified in Chapter 4. Apart from the physical feasibility of cover crops in rainfed orchards, there is also a lack of knowledge about the economic feasibility. In order to fill this knowledge gap, there is a need for field studies that include a cost-benefit analysis.

It was illustrated in Chapter 5 that the quantification and modelling of hydrological connectivity is still in its infancy. The observed relation between the integrated connectivity scale and storm magnitude needs to be confirmed by additional, independent observations. The next step would be to develop and improve rainfall-runoff models for semi-arid agricultural catchments by incorporating the connectivity theory and observed empirical relations. Another line of research would be to develop techniques for mapping spatial patterns in soil moisture and pathways of overland flow over large areas.

The results of Chapter 6 are still somewhat preliminary, and can be improved in a number of ways. The field experiment will continue for another year, so that the observations and yield data become more reliable. Furthermore, the quality of the results can be improved by increasing the number of moisture sensors: the present study has only two sensors per treatment. It will also be helpful to compare the cover crop and tillage treatment in terms of gross margin. The input data of the water balance model can be improved as well.

BIBLIOGRAPHY

Agenbag, G.A., Maree, P.C.J., 1991. Effect of tillage on some soil properties, plant development and yield of spring wheat (Triticum aestivum L.) in stony soil. Soil & Tillage Research 21, 97-112.

Allard, D., 1994. Simulating a geological lithofacies with respect to connectivity information using the truncated Gaussian model. Proceedings of the Geostatistical Simulation Workshop, Fontainbleau, France, 27-28 May 1993. Kluwer Acad., Norwell, Mass., pp. 197-211.

Allen, R.G., Pereira, L.S., Raes, D., Smith, M., 1998. Crop evapotranspiration - guidelines for computing crop water requirements. FAO, Rome.

Alvarado, M., Civantos, M., Duran, J.M., 2004. Plagas. In: Barranco, D., Fernández-Escobar, R., Rallo, L. (Eds.), El cultivo del olivo. Ediciones Mundi-Prensa, Madrid, pp. 481-556.

AMINAL, 2002a. Wegwijzer doorheen het erosiebesluit. Ministerie van de Vlaamse Gemeenschap, administratie Landbeheer, Brussel.

AMINAL, 2002b. Kleinschalige erosiebestrijdingswerken: een praktijkvoorbeeld. Ministerie van de Vlaamse Gemeenschap, administratie Landbeheer, Brussel.

AMINAL, 2002c. Werk maken van erosiebestrijding. Ministerie van de Vlaamse Gemeenschap, administratie Landbeheer, Brussel.

Amoros, C., Bornette, G., 2002. Connectivity and biocomplexity in waterbodies of riverine floodplains. Freshwater Biology 47, 761-776.

Arnáez, J., Larrea, V., Ortigosa, L., 2004. Surface runoff and soil erosion on unpaved forest roads from rainfall simulation tests in northeastern Spain. Catena 57, 1-14.

Arshad, M.A., Gill, K.S., 1997. Barley, canola and wheat production under different tillage-fallow-green manure combinations on a clay soil in a cold, semiarid climate. Soil & Tillage Research 43, 263-275.

Austin, R., Playán, E., Gimeno, J., 1998. Water storage in soils during the fallow: prediction of the effects of rainfall pattern and soil conditions in the Ebro valley of Spain. Agricultural Water Management 36, 213-231.

Beaufoy, G., 2003. The environmental impact of olive oil production in the European Union: Practical options for improving the environmental impact. European Forum on Nature Conservation and Pastoralism, Peterborough. http://ec.europa.eu/environment/agriculture/pdf/oliveoil.pdf. Last viewed on 3-8-2008.

Beede, R.H., Goldhamer, D.A., 1994. Olive irrigation management. In: Ferguson, L., Sibbett, G.S., Martin, G.C. (Eds.), Olive production manual. University of California, Division of Agriculture and Natural Resources, pp. 61-68.

Bellin, N., van Wesemael, B., Meerkerk, A., Vanacker, V., Barbera, G.G., Submitted. The decay of soil and water conservation structures - A case study from south east Spain. Catena.

Beven, K.J., 2001. Rainfall-runoff modelling. Wiley & Sons, Chichester.

Boardman, J., 2006. Soil erosion science: Reflections on the limitations of current approaches. Catena 68, 73-86.

Borselli, L., Torri, D., Poesen J., Salvador Sanchis, P., 2001. Effects of water quality on infiltration, runoff and interrill erosion processes during simulated rainfall. Earth Surface Processes and Landforms 26, 329-342.

Borselli, L., Pellegrini, S., Torri, D., Bazzoffi, P., 2002. Tillage erosion and land levelling: evidences in Tuscany (Italy). In: Rubio, J., Morgan, R., Asins, S., Andreu, V. (Eds.), Man and soil at the third millenium. Geoforma Ediciones - CIDE, Logroño, pp. 1341-1350.

Borselli, L., Torri, D., Øygarden, L., De Alba, S., Martínez-Casasnovas, J.A., Bazzoffi, P., Jakab, G. 2006. Land levelling. In: Boardman J., Poesen J., (Eds.), Soil erosion in Europe. John Wiley & Sons, Ltd, pp. 643-658.

Bowyer-Bower, T.A.S., Burt, T.P., 1989. Rainfall simulators for investigating soil response to rainfall. Soil Technology 2, 1-16.

Bracken, L.J, Croke, J., 2007. The concept of hydrological connectivity and its contribution to understanding runoff dominated geomorphic systems. Hydrological Processes 21, 1749-1763.

Bueno, J., Amiama, C., Hernanz, J.L., 2007. No-tillage drilling of Italian ryegrass (Lolium multiflorum L.): Crop residue effects, yields and economic benefits. Soil & Tillage Research 95, 61-68.

Caballero, R., 2001. Typology of cereal-sheep farming systems in Castile-La Mancha (south-central Spain). Agricultural Systems 68, 215-232.

Cammeraat, E.L.H., 2004. Scale dependent thresholds in hydrological and erosion response of a semi-arid catchment in southeast Spain. Agriculture, Ecosystems and Environment 104, 317-332.

Campbell, A., 1994. Community first: landcare in Australia. International institute for environment and development - Gatekeeper series No. 42, London.

Cantero-Martínez, C., Angas, P., Lampurlanés, J., 2003. Growth, yield and water productivity of barley (Hordeum vulgare L.) affected by tillage and N fertilization in Mediterranean semiarid, rainfed conditions of Spain. Field Crops Research 84, 341-357.

Cantón, Y., Solé-Benet, A., Domingo, F., 2004. Temporal and spatial patterns of soil moisture in semiarid badlands of SE Spain. Journal of Hydrology 285, 199-214.

Castillo, V., Barberá, G.G., Mosch, W., Navarro-Cano, J.A., Conesa, C., López-Bermúdez, F., 2002. Seguimiento y evaluación de los trabajos de restauración hidrológico-forestal. In: López Bermúdez, F. (Eds.), Siguimiento y evaluación de los efectos sobre el medio natural de la sequía y los procesos erosivos en la Región de Murcia. Consejería de Agricultura, Agua y Medio Ambiente de la Región de Murcia, Murcia, pp. 166-233.

Castillo, V.M., Gómez-Plaza, A., Martínez-Mena, M., 2003. The role of antecedent soil water content in the runoff response of semiarid catchments: a simulation approach. Journal of Hydrology 284, 114-130.

Castillo, V.M., Mosch, W.M., Conesa García, C., Barberá, G.G., Navarro Cano, J.A., López-Bermúdez, F., 2007. Effectiveness and geomorphological impacts of check dams for soil erosion control in a semiarid Mediterranean catchment: El Cárcavo (Murcia, Spain). Catena 70, 416-427.

Castro, G., Romero, P., Gómez, J.A., Fereres, E., 2006. Rainfall redistribution beneath an olive orchard. Agricultural Water Management 86, 249-258.

Catlin, P.B., 1996. Root systems and root physiology. In: Micke, W.C. (Eds.), Almond production manual. University of California, Division of Agriculture and Natural resources, Oakland, pp. 107-112.

Caylor, K.K., Shugart, H.H., Dowty, P.R., Smith, T.M., 2003. Tree spacing along the Kalahari transect in southern Africa. Journal of Arid Environments 54, 281-296.

Ceballos, A., Schnabel, S., 1998. Hydrological behaviour of a small catchment in the dehesa landuse system (Extremadura. SW Spain). Journal of Hydrology 210, 146-160.

Cerdà, A., 1997. Soil erosion after land abandonment in a semiarid environment of Southeastern Spain. Arid Soil Research and Rehabilitation 11, 163-176.

Cheema, Z.A., Khaliq, A., 2000. Use of sorghum allelopathic properties to control weeds in irrigated wheat in a semi arid region of Punjab. Agriculture, Ecosystems & Environment 79, 105-112.

Christiansen, S., Bounejmate, M., Bahhady, F., Thomson, E., Mawlawi, B., Singh, M., 2000. On-farm trials with forage legume-barley compared with fallow-barley rotations and continuous barley in north-west Syria. Experimental Agriculture 36, 195-204.

Clarke, M.L., Rendell, H.M., 2000. The impact of the farming practice of remodelling hillslope topography on badland morphology and soil erosion processes. Catena 40, 229-250.

Costagli, G., Gucci, R., Rapoport, H.F., 2003. Growth and development of fruits of olive 'Frantoio' under irrigated and rainfed conditions. Journal of Horticultural Science & Biotechnology 78, 119-124.

CREM (Centro Regional de Estadística de Murcia), 2004. Anuario estadístico de la Región de Murcia. 2004. Tomo II : Datos Municipales. http://www.carm.es/econet/anuario/a04/anuario.html. Last viewed on 28-08-2008.

CREM (Centro Regional de Estadística de Murcia), 2006. Anuario estadístico de la Región de Murcia 2006 - Tomo II: Datos Municipales. http://www.carm.es/econet/anuario/a06/anuario.html. Last viewed on 28-08-2008.

Croke, J., Mockler, S., Fogarty, P., Takken, I., 2005. Sediment concentration changes in runoff pathways from a forest road network and the resultant spatial pattern of catchment connectivity. Geomorphology 68, 257-268.

De Alba, S., 2003. Simulating long-term soil redistribution generated by different patterns of mouldboard ploughing in landscapes of complex topography. Soil & Tillage Research 71, 71-86.

De Baets, S., Poesen, J., Gyssels, G., Knapen, A., 2006. Effects of grass roots on the erodibility of topsoils during concentrated flow. Geomorphology 76, 54-67.

De Baets, S., Poesen, J., Knapen, A., Barberá, G.G., Navarro, J.A., 2007. Root characteristics of representative Mediterranean plant species and their erosion-reducing potential during concentrated runoff. Plant and Soil 294, 169-183.

De Graaff, J., Eppink, L.A.A.J., 1999. Olive oil production and soil conservation in southern Spain, in relation to EU subsidy policies. Land Use Policy 16, 259-267.

De Luna, E., Laguna, A., Giráldez, J.V., 2000. The role of olive trees in rainfall erosivity and runoff and sediment yield in the soil beneath. Hydrology and Earth System Sciences 4, 141-153.

Debussche, M., Lepart, J., Dervieux, A., 1999. Mediterranean landscape changes: evidences from old postcards. Global Ecology and Biogeography 8, 3-15.

Denison, R.F., Bryant, D.C., Kearney, T.E., 2004. Crop yields over the first nine years of LTRAS, a long-term comparison of field crop systems in a Mediterranean climate. Field Crops Research 86, 267-277.

Díaz-Ambrona, C.H., Mínguez, I.M., 2001. Cereal-legume rotations in a Mediterranean environment: biomass and yield production. Field Crops Research 70, 139-151.

Doolittle, W.E., 1983. Agricultural expansion in a marginal area of Mexico. Geographical Review 73, 301-313.

Doorenbos, J., Pruitt, W.O., 1977. FAO irrigation and drainage paper 24 - Crop water requirements. FAO, Rome.

EC (European Commission, Directorate-general for agriculture), 2002. The olive oil sector in the European Union. KF-44-02-084-EN-D.

EC (European Commission, Directorate-general for agriculture and rural development), 2007. The common agricultural policy explained. KF-77-07-075-EN-D.

Edstrom, J.P., Micke, W.C., 1998. Dual variety rows increase almond yields. Second International Symposium on Pistachios and Almonds. Leuven, pp. 469-472.

Elwell, H.A., Stocking, M.A., 1976. Vegetal cover to estimate soil erosion hazard in Rhodesia. Geoderma 15, 61-70.

Ennabli, N., 1993. Les aménagements hydrauliques et hydro-agricoles en Tunisie. Institut national agronomique de Tunis. Département du génie rural des eaux et des forêts, Tunis.

EU, 2002. Communication from the Commission to the Council and the European Parliament - Mid-Term Review of the Common Agricultural Policy (DOCI 52002DC0394).

European Commission, 2007. Conditions for restoration & mitigation of desertified areas using vegetation (RECONDES) - Review of literature and present knowledge. European Commission, Brussels. ISBN 92-79-03072-8.

Ewert, F., Rounsevell, M.D.A., Reginster, I., Metzger, M.J., Leemans, R., 2005. Future scenarios of European agricultural land use: I. Estimating changes in crop productivity. Agriculture, Ecosystems and Environment 117, 101-116.

FAO, 1990. FAO-Unesco soil map of the world. Revised legend. Soils bulletin 60. FAO, Rome.

Faulkner, H., 1995. Gully erosion associated with the expansion of unterraced almond cultivation in the coastal Sierra de Lujar, S. Spain. Land Degradation and Rehabilitation 6, 179-200.

Faulkner, H., Ruiz, J., Zukowskyj, P., Downward, S., 2003. Erosion risk associated with rapid and extensive agricultural clearances on dispersive materials in southeast Spain. Environmental Science and Policy 6, 115-127.

Ferguson, L., Sibbett, G., Martin, G., 1994. Olive production manual. University of California, Division of Agriculture and Natural Resources, Oakland.

Fiener, P., Auerswald, K., 2003. Effectiveness of grassed waterways in reducing runoff and sediment delivery from agricultural watersheds. Journal of Environmental Quality 32, 927-936.

Fiener, P., Auerswald, K., 2005. Measurement and modeling of concentrated runoff in grassed waterways. Journal of Hydrology 301, 198-215.

Fleskens, L., 2007. Conservation scenarios for olive farming on sloping land in the Mediterranean. PhD thesis, Wageningen University.

Fleskens, L., Stroosnijder, L., 2007. Is soil erosion in olive groves as bad as often claimed? Geoderma 141, 260-271.

Frot, E., van Wesemael, B., Benet, A.S., House, M.A., 2008. Water harvesting potential in function of hillslope characteristics, a case study from the Sierra de Gador (Almeria province, southeast Spain). Journal of Arid Environments 72, 1213-1231.

Gale, W.J., McColl, R.W., Fang, X., 1993. Sandy fields traditional farming for water conservation in China. Journal of Soil and Water Conservation 78, 474-477.

Gálvez, M., Parra, M.A., Navarro, C., 2004. Relating tree vigour to the soil and landscape characteristics of an olive orchard in a marly area of southern Spain. Scientia Horticulturae 101, 291-303.

Garcia, C., Hernández, T., 2004. Indicators and thresholds for desertification, soil quality and remediation - site description. Centro de Edafología y Biología Aplicada del Segura (CEBAS-CSIC), Murcia.

Ghrab, M., Mimoun, M.B., Triki, H., Hellali, R., 2002. Yield of twenty four almond cultivars in a dry area climate in Tunisia: five years of study. Third International Symposium on Pistachios and Almonds. Leuven.

Giráldez, J.V., Ayuso, J.L., Garcia, A., López, J.G., Roldán, J., 1988. Water harvesting strategies in the semiarid climate of southeastern Spain. Agricultural Water Management 14, 253-563.

Godini, A., Palasciano, M., 1998. Growth and yield of four self-fruitful and four self-unfruitful almonds on three rootstocks: a thirteen year study. Second International Symposium on Pistachios and Almonds. Leuven, pp. 200-207.

Goldhamer, D.A., 1996. Irrigation scheduling. In: Micke, W.C. (Eds.), University of California, Division of Agriculture and Natural Resources, pp. 171-178.

Gómez, J.A., Giráldez, J.V., Pastor, M., Fereres, E., 1999. Effects of tillage method on soil physical properties, infiltration and yield in an olive orchard. Soil & Tillage Research 52, 167-175.

Gómez, J.A., Giráldez, J.V., Fereres, E., 2001. Rainfall interception by olive trees in relation to leaf area. Agricultural Water Management 49, 65-76.

Gómez, J.A., Vanderlinden, K., Giráldez, J.V., Fereres, E., 2002. Rainfall concentration under olive trees. Agricultural Water Management 55, 53-70.

Gómez, J.A., Romero, P., Giráldez, J.V., Fereres, E., 2004. Experimental assessment of runoff and soil erosion in an olive grove on a Vertic soil in southern Spain as affected by soil management. Soil Use and Management 20, 426-431.

Gómez, J.A., Giráldez, J.V., Fereres, E., 2005. Water erosion in olive orchards in Andalusia (Southern Spain): a review. Geophysical Research Abstracts 7, 08406.

Gómez-Plaza, A., Martínez-Mena, M., Albaladejo, J., Castillo, V.M., 2001. Factors regulating spatial distribution of soil water content in small semiarid catchments. Journal of Hydrology 253, 211-226.

Google Earth, 2006. http://earth.google.com/. Last viewed on 28-08-2008.

Groenevelt, P.H., Van Straaten, P., Rasiah, V., Simpson, J., 1989. Modifications in evaporation parameters by rock mulches. Soil Technology 2, 279-285.

Grove, A.T., Rackham, O., 2003. The nature of Mediterranean Europe - An ecological history. Yale University Press.

Hargreaves, G.H., Samani, Z.A., 1985. Reference crop evapotranspiration from temperature. Applied Engineering in Agriculture 1, 96-99.

Hartwig, N.L., Ammon, H.U., 2002. Cover crops and living mulches. Weed Science 50, 688-699.

Harvey, A.M., 2002. Effective timescales of coupling within fluvial systems. Geomorphology 44, 175-201.

Hendricks, L.C., 1996. Orchard planning, design and development. In: Micke, W.C. (Eds.), Almond production manual. University of California, Division of Agriculture and Natural Resources, Oakland, pp. 47-51.

Hernández, A.J., Lacasta, C., Pastor, J., 2005. Effects of different management practices on soil conservation and soil water in a rainfed olive orchard. Agricultural Water Management 77, 232-248.

Hooke, J., 2003. Coarse sediment connectivity in river channel systems: a conceptual framework and methodology. Geomorphology 56, 79-94.

Hooke, J.M., Sandercock, P.J. (Eds.), 2007. RECONDES: Conditions for restoration and mitigation of desertified areas in Southern Europe using vegetation - Final report.

Hudson, N., 1985. Soil conservation. Batsford academic & educ., London.

Hutchinson, M.F., 1989. A new procedure for gridding elevation and stream line data with automatic removal of spurious pits. Journal of Hydrology 106, 211-232.

ICONA, 1986. Mapa de suelos escala 1:100.000 Cieza - 891. Ministerio de Agricultura, Pesca y Alimentación, Madrid.

Ingels, C.A., Bugg, R.L., McGourty, G.T., Christensen, L.P., 1998. Cover cropping in vineyards - a growers handbook. University of California, Division of Agriculture and Natural Resources, Oakland.

IPCC (Intergovernmental Panel on Climate Change), 2007. Climate Change 2007: Synthesis Report. http://www.ipcc.ch/pdf/assessment-report/ar4/syr/ar4_syr.pdf. Last viewed on 04-08-2008.

Islam, N., Wallender, W.W., Mitchell, J., Wicks, S., Howitt, R.E., 2006. A comprehensive experimental study with mathematical modeling to investigate the affects of cropping practices on water balance variables. Agricultural Water Management 82, 129-147.

Joffre, R., Vacher, J., Delosllanos, C., Long, G., 1988. The dehesa - an agrosilvopastoral system of the Mediterranean region with special reference to the Sierra Morena of Spain. Agroforestry systems 6, 71-96.

Kirpich, T., 1940. Time of concentration of small agricultural watersheds. Journal of Civil Engineering 10, 362.

Koorevaar, P., Menelik, G., Dirksen, C., 1983. Elements of soil physics. Elsevier Science Publishers B.V., Amsterdam.

Kosmas, C., Danalatos, N., Cammeraat, L.H., Chabart, M., Diamantopoulos, J., Farand, R., Gutierrez, L., Jacob, A., Marques, H., Martinez-Fernandez, J., Mizara, A., Moustakas, N., Nicolau, J.M., Oliveros, C., Pinna, G., Puddu, R., Puigdefabregas, J., Roxo, M., Simao, A., Stamou, G., Tomasi, N., Usai, D., Vacca, A., 1997. The effect of land use on runoff and soil erosion rates under Mediterranean conditions. Catena 29, 45-59.

Koulouri, M., Giourga, C., 2007. Land abandonment and slope gradient as key factors of soil erosion in Mediterranean terraced lands. Catena 69, 274-281.

Lambin, E.F., Rounsevell, M.D.A., Geist, H.J., 2004. Are agricultural land-use models able to predict changes in land-use intensity? Agriculture, Ecosystems and Environment 82, 321-331.

Lampurlanés, J., Angás, P., Cantero-Martínez, C., 2002. Tillage effects on water storage during fallow, and on barley root growth and yield in two constrasting soils of the semi-arid Segarra region in Spain. Soil & Tillage Research 65, 207-220.

Landry, D., Dousset, S., Fournier, J.C., Andreux, F. 2005. Leaching of glyphosate and AMPA under two soil management practices in Burgundy vineyards (Vosne-Romanée, 21-France). Environmental Pollution 138, 191-200.

Lennartz, B., Louchart, X., Voltz, M., Andreux, P., 1997. Diuron and simazine losses to runoff water in Mediterranean vineyards. Journal of Environmental Quality 26, 1493-1502.

Leonard, J., Andrieux, P., 1998. Infiltration characteristics of soils in Mediterranean vineyards in southern France. Catena 32, 209-223.

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

Li, X., 2003. Gravel-sand mulch for soil and water conservation in the semiarid loess region of northwest China. Catena 52, 105-127.

Lindstrom, M.J., Nelson, W.W., Schumacher, T.E., 1992. Quantifying tillage erosion rates due to mouldboard ploughing. Soil & Tillage Research 22, 91-98.

López, M.V., Arrúe, J.L., 1997. Growth, yield and water use efficiency of winter barley in response to conservation tillage in a semi-arid region of Spain. Soil & Tillage Research 44, 35-54.

López, M.V., Moret, D., Gracia, R., Arrúe, J.L., 2003. Tillage effects on barley residue cover during fallow in semiarid Aragon. Soil & Tillage Research 72, 53-64.

López-Granados, F., Jurado-Expósito, M., Álamo, S., García-Torres, L., 2004. Leaf nutrient variability and site-specific fertilization maps within olive (Olea europaea L.) orchards. European Journal of Agronomy 21, 209-222.

Lövenstein, H., Lantinga, E.A., Rabbinge, R., Van Keulen, H., 1995. Principles of production ecology. Wageningen Agricultural University, Department of theoretical production ecology, Wageningen.

MacDonald, L.H., Sampson, R.W., Anderson, D.M., 2001. Runoff and erosion at the plot and road segment scales, St John, US Virgin Islands. Earth Surface Processes and Landforms 26, 251-272.

MAPA (Ministerio de Agricultura, Pesca y Alimentación), 2004. Anuario de Estadística Agroalimentaria. http://www.mapa.es/es/estadistica/pags/anuario/Anu_04/indice.asp.

Martínez-Casasnovas, J.A., Sánchez-Bosch, I., 2000. Impact assessment of changes in land use/conservation practices on soil erosion in the Penedès-Anoia vineyard region (NE Spain). Soil & Tillage Research 57, 101-106.

Mas, M.T., Verdú, A.M.C., 2003. Tillage system effects on weed communities in a 4-year crop rotation under Mediterranean dryland conditions. Soil & Tillage Research 74, 15-24.

Matthias, A.D., Fimbres, A., Sano, E.E., Post, D.F., Accioly, L., Batchily, A.K., Ferreira, L.G., 2000. Surface roughness effects on soil albedo. Soil Science Society of America Journal 64, 1035-1041.

MEA (Millenium Ecosystem Assessment), 2005. Ecosystems and human well-being: desertification synthesis. World Resources Institute, USA.

Meerkerk, A.L., van Wesemael, B., Cammeraat, E., Accepted. Water availability in almond orchards on marl soils in south east Spain: The role of evaporation and runoff. Journal of Arid Environments.

Mellouli, H.J., Van Wesemael, B., Poesen, J., Hartmann, R., 2000. Evaporation losses from bare soils as influenced by cultivation techniques in semi-arid regions. Agricultural Water Management 42, 355-369.

Metochis, C., Orphanos, P.I., 1997. Yield of barley under Mediterranean conditions of variable rainfall. Agricultural and Forest Meteorology 85, 251-258.

Michelakis, N., Vouyoukalou, E., Clapaki, G., 1996. Water use and soil moisture depletion by olive trees under different irrigation conditions. Agricultural Water Management 29, 315-325.

Micke, W.C., 1996. Almond production manual. University of California, Division of Agriculture and Natural resources, Oakland.

Mishra, S.K., Singh, V.P., 2003. Soil Conservation Service curve number (SCS-CN) methodology. Kluwer Academic Publishers, Dordrecht.

Mitchell, T.D., Hulme, M., New, M., 2002. Climate data for political areas. Area 34, 109-112.

Morgan, R.P.C., 1995. Soil erosion and conservation. Longman Group Limited, London.

Moriana, A., Orgaz, F., Pastor, M., Fereres, E., 2003. Yield responses of a mature olive orchard to water deficits. Journal of the American Society for Horicultural Sciene 128, 425-431.

Mueller, E.N., Wainwright, J., Parsons, A.J., 2007. Impact of connectivity on the modelling of overland flow within semiarid shrubland environments. Water Resources Research 43, W09412.

Mulligan, M., 1998. Hydrology in a varying as well as changing environment. Hydrology in a changing environment. Volume II. British Hydrological Society.

Mygdakos, E., Avgoulas, C., Bilalis, D., 2005. Conventional, reduced tillage and no-tillage systems for cotton growing under Mediterranean conditions: a cultural and economic approach. Journal of Food, Agriculture & Environment 3, 173-179.

Nakicenovic, N., Alcamo, J., Davis, G., De Vries, B., Fenhann, J., Gaffin, S., Gergory K.; Grübler, Grübler, A., Jung, T.Y., Kram. T.; Emilio la Rovere, Emilio la Rovere, E., Michaelis, L., Mori, S., Morita, T., Pepper, W., Pitcher, H., Price, L., Riahi, K., Roehrl, A., et al., 2000. Special report on emission scenarios: a special report of working group III of the intergovernmental panel on climate change. Cambridge University Press, Cambridge.

Nasri, S., Albergel, J., Cudennec, C., 2004. Hydrological processes in macrocatchment water harvesting in the arid region of Tunisia: the traditional system of tabias. Hydrological Sciences Journal 49, 261-272.

Navarro, C., Parra, M.A., 2004. Plantación. In: Barranco, D., Fernández-Escobar, R., Rallo, L. (Eds.), El cultivo del olivo. Ediciones Mundi-Prensa, Madrid, pp. 185-228.

New, M., Hulme, M., Jones, P., 1999. Representing twentieth-century space-time climate variability, Part I: Development of a 1961-90 mean monthly terrestrial climatology. Journal of Climate 12, 829-856.

Nicolau, J.M., Solé-Benet, A., Puigdefábregas, J., Gutiérrez, L., 1996. Effects of soil and vegetation on runoff along a catena in semi-arid Spain. Geomorphology 14, 297-309.

Nuberg, I., Yunusa, I., 2003. Olive water use and yield - monitoring the relationship. Rural Industries Research and Development Corporation, Barton, Australia.

Ocampo, C.J., Sivapalan, M., Oldham, C., 2006. Hydrological connectivity of upland-riparian zones in agricultural catchments: Implications for runoff generation and nitrate transport. Journal of Hydrology 331, 643-658.

Olesen, J.E., Bindi, O., 2002. Consequences of climate change for European agricultural productivity, land use and policy. European Journal of Agronomy 16, 239-262.

Oñate, J.J., Peco, B., 2005. Policy impact on desertification: stakeholders' perceptions in southeast Spain. Land Use Policy 22, 103-114.

Oostwoud Wijdenes, D., Poesen, J., Vandekerckhove, L., De Luna, E., 1997. Chiselling effects on the vertical distribution of rock fragments in the tilled layer of a Mediterranean soil. Soil & Tillage Research 44, 55-66.

Oostwoud Wijdenes, D.J., Poesen, J., Vandekerckhove, L., Ghesquiere, M., 2000. Spatial distribution of gully head activity and sediment supply along an ephemeral channel in a Mediterranean environment. Catena 39, 147-167.

Orgaz, F., Fereres, E., 2004. Riego. In: Barranco, D., Fernández-Escobar, R., Rallo, L. (Eds.), El cultivo del olivo. Ediciones Mundi-Prensa, Madrid, pp. 321-346.

Oweis, T., Zhang, H., Pala, M., 2000. Water use efficiency of rainfed and irrigated bread wheat in a Mediterranean environment. Agronomy Journal 92, 231-238.

Pala, M., Harris, H.C., Ryan, J., Makboul, R., Dozom, S., 2000. Tillage systems and stubble management in a Mediterranean-type environment in relation to crop yield and soil moisture. Experimental Agriculture 36, 223-242.

Palomo, M.J., Moreno, F., Fernández, J.E., Díaz-Espejo, A., Girón, I.F., 2002. Determining water consumption in olive orchards using the water balance approach. Agricultural Water Management 55, 15-35.

Pastor, M., 2004. Sistemas de manejo del suelo. In: Barranco, D., Fernández-Escobar, R., Rallo, L. (Eds.), El cultivo del olivo. Ediciones Mundi-Prensa, Madrid, pp. 229-285.

Pinto-Correia, T., Mascarenhas, J., 1999. Contribution to the extensification/intensification debate: new trends in the Portuguese montado. Landscape and Urban Planning 46, 125-131.

Pinto-Correia, T., 2000. Future development in Portuguese rural areas: how to manage agricultural support for landscape conservation? Landscape and Urban Planning 50, 95-106.

Poesen, J.W.A., Hooke, J.M., 1997. Erosion, flooding and channel management in Mediterranean environments of southern Europe. Progress in Physical Geography 21, 157-199.

Poesen, J., Van Wesemael, B., Govers, G., Martinez-Fernandez, J., Desmet, P., Vandaele, K., Quine, T., Degraer, G., 1997. Patterns of rock fragment cover generated by tillage erosion. Geomorphology 18, 183-197.

Post, D.F., Fimbres, A., Matthias, A.D., Sano, E.E., Accioloy, L., Batchily, A.K., Ferreira, L.G., 2000. Predicting soil albedo from soil color and spectral reflectance data. Soil Science Society of America Journal 61, 1027-1034.

Pringle, C., 2003. What is hydrological connectivity and why is it ecologically important? Hydrological Processes 17, 2685-2689.

Puigdefabregas, J., del Barrio, G., Boer, M.M., Gutiérrez, L., Solé, A., 1998. Differential responses of hillslope and channel elements to rainfall events in a semi-arid area. Geomorphology 23, 337-351.

Raes, D., 2002. BUDGET - A soil water and salt balance model. Institute for Land and Water Management, K.U. Leuven, Leuven.

Raes, D., Geerts, S., Kipkorir, E., Wellens, J., Sahli, A., 2006. Simulation of yield decline as a result of water stress with a robust soil water balance model. Agricultural Water Management 81, 335-357.

Rallo, L., Cuevas, J., 2004. Fructificatión y producción. In: Barranco, D., Fernández-Escobar, R., Rallo, L. (Eds.), El cultivo del olivo. Ediciones Mundi-Prensa, Madrid, pp. 125-158.

Ramos, M.C., Mulligan, M., 2005. Spatial modelling of the impact of climate variability on the annual soil moisture regime in a mechanized Mediterranean vineyard. Journal of Hydrology 306, 287-301.

Ramos, M.C., Martínez-Casasnovas, J.A., 2006. Impact of land levelling on soil moisture and runoff variability in vineyards under different rainfall distribution in a Mediterranean climate and its influence on crop productivity. Journal of Hydrology 321, 131-146.

Rana, G., Katerji, N., 2000. Measurement and estimation of actual evapotranspiration in the field under Mediterranean climate: a review. European Journal of Agronomy 13, 125-153.

Reaney, S.M., Bracken, L.J., Kirkby, M.J., 2007. Use of the connectivity of runoff model (CRUM) to investigate the influence of storm characteristics on runoff generation and connectivity in semi-arid areas. Hydrological Processes 21, 894-906.

RECONDES, 2007. Combating land degradation by minimal intervention: The connectivity reduction approach. University of Portsmouth.

Revel, J.C., Coste, N., Cavalie, J., Costes, J.L., 1989. Permiers résultats expérimentaux sur l'entrainement mecanique des terres par le travail du sol dans le Terrefort Toulousain (France). Cahiers ORSTOM, Série Pédologie 25, 111-118.

Ritchie, J.T., 1972. Model for predicting evaporation from a row crop with incomplete cover. Water Resources Research 8, 1204-1213.

Rockström, J., Valentin, C., 1997. Hillslope dynamics of on-farm generation of surface water flows: The case of rain-fed cultivation of pearl millet on sandy soil in the Sahel. Agricultural Water Management 33, 183-210.

Rockström, J., Jansson, P., Barron, J., 1998. Seasonal rainfall partitioning under runon and runoff conditions on sandy soil in Niger. On-farm measurements and water balance modelling. Journal of Hydrology 210, 68-92.

Rockström, J., 2003. Resilience building and water demand management for drought mitigation. Physics and Chemistry of the Earth 28, 869-877.

Romero, P., Navarro, J.M., García, F., Ordaz, P.B., 2004. Effects of regulated deficit irrigation during the pre-harvest period on gas exchange, leaf development and crop yield of mature almond trees. Tree Physiology 24, 303-312.

Rosato, P., Giupponi, C., 2003. What future for Mediterranean agriculture? In: Giupponi, C., Shechter, M. (Eds.), Climate change in the Mediterranean. Edward Elgar, pp. 133-158.

Rounsevell, M.D.A., Ewert, F., Reginster, I., Leemans, R., Carter, T.R., 2005. Future scenarios of European agricultural land use II. Projecting changes in cropland and grassland. Agriculture, Ecosystems and Environment 117, 117-135.

Ruecker, G., Schad, P., Alcubilla, M.M., Ferrer, C., 1998. Natural regeneration of degraded soils and site changes on abandoned agricultural terraces in Mediterranean Spain. Land Degradation & Development 9, 179-188.

Russi, L., Falcinelli, M., 1999. Characterization and potential role of annual clover landraces in the farming systems of Mediterranean areas. Journal of Agricultural Science 132, 387-397.

Sala, O.E., Parton, W.J., Joyce, L.A., Lauenroth, W.K., 1988. Primary production of the central grassland region of the United States. Ecology 69, 40-45.

Sánchez-Girón, V., Serrano, A., Hernanz, J.L., Navarrete, L., 2004. Economic assessment of three long-term tillage systems for rainfed cereal and legume production in semiarid central Spain. Soil & Tillage Research 78, 35-44.

Sene, K.J., 1996. Meteorological estimates for the water balance of a sparse vine crop growing in semiarid conditions. Journal of Hydrology 179, 259-280.

Shaw, E.M., 1994. Hydrology in practice. Chapman & Hall, London.

Simpson, P.G., Siddique, K.H.M., 1994. Soil type influences relative yield of barley and wheat in a Mediterranean-type environment. Journal of Agronomy & Crop Science 172, 147-160.

Snyder, R.L., Connell, J.H., 1996. Frost protection. In: Micke, W.C. (Eds.), Almond production manual. University of California, Division of Agriculture and Natural Resources, Oakland, pp. 155-166.

Soil Conservation Service, 1985. National Engineering Handbook: section 4, Hydrology. U.S. department of agriculture, Washington D.C.

Soini, E., 2005. Land use change patterns and livelihood dynamics on the slopes of Mt. Kilimanjaro, Tanzania. Agricultural Systems 85, 306-323.

Stieglitz, M., Shaman, J., McNamara, J., Engel, V., Shanley, J., Kling, G.W., 2003. An approach to understanding hydrologic connectivity on the hillslope and the implications for nutrient transport. Global Biogeochemical Cycles 17, 1105.

Taillefumier, F., Piégay, H., 2003. Contemporary land use changes in prealpine Mediterranean mountains: a multivariate GIS-based approach applied to two municipalities in the Southern French Prealps. Catena 51, 267-296.

Tjhie, A. 1991. The competition for water between almond trees and a cover crop: a simulation study.

Tonnito, C., David, M.B., Drinkwater, L.E., 2006. Replacing bare fallows with cover crops in fertilizer-intensive cropping systems: A meta-analysis of crop yield and N dynamics. Agriculture, Ecosystems and Environment 112, 58-72.

Topp, C.F.E., Mitchell, M., 2003. Forecasting the environmental and socioeconomic consequences of changes in the Common Agricultural Policy. Agricultural Systems 76, 227-252.

Torri, D., Borselli, L., Calzolari, C., Yañez, M.S., Salvador Sanchis, M.P., 2002. Soil erosion, land use, soil qualities and soil functions: effects of erosion. In: Rubio, J., Morgan, R., Asins, S., Andreu, V. (Eds.), Man and soil at the third millenium. Geoforma Ediciones - CIDE, Logroño, pp. 131-148.

Trambouze, V., Voltz, M., 2001. Measurement and modelling of the transpiration of a Mediterranean vineyard. Agricultural and Forest Meteorology 107, 153-166.

Trapero, A., Blanco, M.A., 2004. Enfermedades. In: Barranco, D., Fernández-Escobar, R., Rallo, L. (Eds.), El cultivo del olivo. Ediciones Mundi-Prensa, Madrid, pp. 557-614.

Tubeileh, A., Bruggeman, A., Turkelboom, F., 2004. Growing olive tree species in marginal dry environments. ICARDA, Aleppo, Syria.

UN-ESA (United Nations, Department of Economic and Social Affairs, Population Division), 2000. Charting the progress of populations, Chapter I: Total population. New York. http://www.un.org/esa/population/publications/charting/3.pdf (last accessed in July 2008).

UNEP, 1997. World atlas of desertification. Arnold, London.

Valentin, C., Bresson, L.-, 1992. Morphology, genesis and classification of surface crusts in loamy and sandy soils. Geoderma 55, 225-245.

Van de Giesen, N.C., Stomph, T.J., de Ridder, N., 2000. Scale effects of Hortonian overland flow and rainfall-runoff dynamics in a West African catena landscape. Hydrological Processes 14, 165-175.

Van Dijk, A.I.J.M., Bruijnzeel, L.A., Rosewell, C.J., 2002. Rainfall intensity - kinetic energy relationships: a critical literature appraisal. Journal of Hydrology 261, 1-23.

Van Oost, K., Govers, G., 2006. Tillage erosion. In: Boardman J., Poesen J., (Eds.), Soil erosion in Europe. John Wiley & Sons, Ltd, pp. 599-608.

van Wesemael, B., Poesen, J., Kosmas, C.S., Danalatos, N.G., Nachtergaele, J., 1996. Evaporation from cultivated soils containing rock fragments. Journal of Hydrology 182, 65-82.

van Wesemael, B., Poesen, J., Solé Benet, L., Cara Barrionuevo, L., Puigdefábregas, J., 1998. Collection and storage of runoff from hillslopes in a semi-arid environment: geomorphic and hydrologic aspects of the aljibe system in Almeria Province, Spain. Journal of Arid Environments 40, 1-14.

van Wesemael, B., Mulligan, M., Poesen, J., 2000. Spatial patterns of soil water balance on intensively cultivated hillslopes in a semi-arid environment: the impact of rock fragments and soil thickness. Hydrological Processes 14, 1811-1828.

van Wesemael, B., Cammeraat, E., Mulligan, M., Burke, S., 2003. The impact of soil properties and topography on drought vulnerability of rainfed cropping systems in southern Spain. Agriculture, Ecosystems and Environment 94, 1-15.

van Wesemael, B., Rambaudt, X., Poesen, J., Muligan, M., Cammeraat, E., Stevens, A., 2006. Spatial patterns of land degradation and their impacts on the water balance of rainfed treecrops: a case study in south east Spain. Geoderma 133, 43-56.

Vandekerckhove, L., Poesen, J., Oostwoud Wijdenes, D., De Figueiredo, T., 1998. Topographical thresholds for ephemeral gully initiation in intensively cultivated areas of the Mediterranean. Catena 33, 271-292.

Vanderlinden, K., Gabriels, D., Giraldez, J.V., 1998. Evaluation of infiltration measurements under olive trees in Cordoba. Soil & Tillage Research 48, 303-315.

Verstraeten, G., Prosser, I.P., Fogarty, P., 2007. Predicting the spatial patterns of hillslope sediment delivery to river channels in the Murrumbidgee catchment, Australia. Journal of Hydrology 334, 440-454.

Villalobos, F.J., Orgaz, F., Testi, L., Fereres, E., 2000. Measurement and modeling of evapotranspiration of olive (Olea europaea L.) orchards. European Journal of Agronomy 13, 155-163.

Wallace, J.S., 2000. Increasing agricultural water use efficiency to meet future food production. Agriculture, Ecosystems and Environment 82, 105-119.

Wemple, B.C., Jones, J.A., Grant, G.E., 1996. Channel network extension by logging roads in two basins, western Cascades, Oregon. Water Resources Bulletin 32, 1195-1207.

Western, A.W., Blöschl, G., Grayson, R.B., 1998. Geostatistical characterisation of soil moisture patterns in the Tarrawarra catchment. Journal of Hydrology 205, 20-37.

Western, A.W., Grayson, R.B., Blöschl, G., Willgoose, G.R., McMahon, T.A., 1999. Observed spatial organization of soil moisture and its relation to terrain indices. Water Resources Research 35, 797-810.

Western, A.W., Blöschl, G., Grayson, R.B., 2001. Toward capturing hydrologically significant connectivity in spatial patterns. Water Resources Research 37, 83-97.

Wortmann, C.S., McIntyre, B.D., Kaizzi, C.K., 2000. Annual soil improving legumes: agronomic effectiveness, nutrient uptake, nitrogen fixation and water use. Field Crops Research 68, 75-83.

Yin, Y.H., Wu, S.H., Zheng, D., Yang, Q.Y., 2005. Regional difference of aridity/humidity conditions change over China during the last thirty years. Chinese Science Bulletin 50, 2226-2233.

Yuan, T., Fengmin, L., Puhai, L., 2003. Economic analysis of rainwater harvesting and irrigation methods, with an example from China. Agricultural Water Management 60, 217-226.

APPENDIX 1 - LIST OF COVER CROPS

List of potential cover crops for semi-arid, rainfed cropping systems.

Species (lat.)	Common name	Type	Relative seedling vigour	Growth habit	Maximum height * (cm)	Time of flowering	Time of maturity	Seedling rate (kg ha^{-1})
System with tillage								
pisum sativum	field pea	legume	mod. to high	viny	46-76	Mar-May	May-Jun	78-112
trifolium alexandrium	berseem clover	legume	mod.	erect	46-76	May-Jun	Jun-Jul	17-22
vicia benghalensis	purple vetch	legume	high	viny	46-61	Apr-May	May-Jun	45-67
vicia faba	bell (fava) bean	legume	high	erect	91-213	Mar-May	May-Jun	112-168
vicia sativa	common vetch	legume	mod.	viny	46-61	Apr-May	May-Jun	45-90
vicia sativa x vicia cordata	Cahaba White' vetch	legume	mod.	viny	46-61	Apr-May	May-Jun	56-90
vicia villosa	hairy vetch	legume	high	viny	46-61	Apr-May	May-Jun	34-56
vicia villosa ssp. dasycarpa	woollypod vetch	legume	high	viny	46-61	Mar-May	Apr-Jun	45-67
avena sativa	oat	grass	high	erect	61-152	Apr-May	May-Jun	112-134
hordeum vulgare	barley	grass	high	erect	61-91	Apr-May	May-Jun	90-112
lolium multiflorum	annual ryegrass	grass	high	erect	91-122	Apr-May	Jun-Aug	22-39
secale cereale cv. 'Merced'	cereal rye	grass	high	erect	91-183	Feb-Apr	Apr-May	67-134

Species (lat.)	Common name	Type	Relative seedling vigour	Growth habit	Maximum height * (cm)	Time of flowering	Time of maturity	Seedling rate (kg ha^{-1})
triticum aestivum	wheat	grass	high	erect	61-102	Apr-May	May-Jun	112-134
x triticosecale	triticale	grass	high	erect	61-152	Apr-May	May-Jun	112-134
brassica spp	brassicas	forb	high	erect	61-152	Mar-May	Apr-Jun	6-13
phacelia tanacetifolia	tansy phacelia	forb	high	semi-erect	30-91	Mar-May	May-Jun	11-17
No-tillage system								
medicago polymorpha	bur medic (burclover)	legume	mod.	prostrate to erect	15-38	Feb-Apr	Apr-May	17-22
trifolium hirtum	rose clover	legume	mod.	erect	20-38	Mar-Apr	May-Jun	17-22
trifolium incarnatum	crimson clover	legume	mod.	erect	30-51	Apr-May	May-Jun	17-28
trifolium subterraneum	subterranean clover	legume	mod.	prostrate to semi-erect	15-38	Mar-May	Apr-Jun	22-28
bromus hordeaceus spp. molliformis	soft chess ('Blando' brome)	grass	high	erect	30-76	Mar-Apr	Apr-May	11-17
vulpia myuros var. hirsuta	foxtail fescue ('Zorro' fescue)	grass	high	erect	30-61	Mar-Apr	Apr	9-13

Source: Ingels et al., 1998. Characteristics apply to the most common varieties used in California and may vary greatly by location and cultivar.

* Maximum height indicates typical height to top of plant or inflorescence in an unmowed monocultural stand.

APPENDIX 2 - OLIVE ORCHARD PROPERTIES AND CLIMATE

Id	Long. (decimal degrees)	Lat. (decimal degrees)	P [a] (mm)	ET_0 [b] (mm)	HI [c] (mm mm^{-1})	Tree density (trees ha^{-1})	Projected canopy cover (m^2 m^{-2})	Source
2	-6.1	37.3	514	1221	0.42	285	0.34	Palomo et al., 2002
3	23.9	35.4	733	1156	0.63	400		Michelakis et al., 1996
3b	23.9	35.4	733	1156	0.63	306	0.20	Google Earth, 2006
3c	23.9	35.4	733	1156	0.63	242	0.27	Google Earth, 2006
3d	23.9	35.4	733	1156	0.63	268	0.23	Google Earth, 2006
4	-4.8	37.6	515	1251	0.41	156	0.20	Gómez et al., 1999
5	-4.8	37.9	486	1247	0.39	277	0.35	Gómez et al., 2002
6	10.2	34.8	279	1638	0.17	17		Ennabli 1993; Tubeileh 2004
6b	10.2	34.8	279	1638	0.17	19	0.05	Google Earth, 2006
6c	10.2	34.8	279	1638	0.17	19	0.03	Google Earth, 2006
6d	10.2	34.8	279	1638	0.17	19	0.02	Google Earth, 2006
7	10.2	35.2	297	1527	0.19	51		Ennabli, 1993
8	10.2	35.6	320	1424	0.22	69		Ennabli, 1993
9	10.0	37.2	567	1092	0.52	156		Ennabli, 1993
10	36.4	35.6	819	1239	0.66	120		A. Tubeileh, 2004 [d]
11	37.1	36.1	370	1276	0.29	120		A. Tubeileh, 2004 [d]
12	35.9	34.9	563	1634	0.34	140		A. Tubeileh, 2004 [d]
13	35.8	35.6	915	1099	0.83	190		A. Tubeileh, 2004 [d]
13b	35.8	35.6	915	1099	0.83	304	0.43	Google Earth, 2006
13c	35.8	35.6	915	1099	0.83	104	0.26	Google Earth, 2006

Id	Long. (decimal degrees)	Lat. (decimal degrees)	P [a] (mm)	ET_0 [b] (mm)	HI [c] (mm mm^{-1})	Tree density (trees ha^{-1})	Projected canopy cover (m^2 m^{-2})	Source
13d	35.8	35.6	915	1099	0.83	82	0.26	Google Earth, 2006
23	10.8	42.9	667	905	0.74	416		Costagli et al., 2003
23b	10.8	42.9	667	905	0.74	334		Google Earth, 2006
23c	10.8	42.9	667	905	0.74	276	0.28	Google Earth, 2006
23d	10.8	42.9	667	905	0.74	283	0.33	Google Earth, 2006
24	-7.4	39.1	642	1092	0.59	90		A. Pinheiro, 2004 [d]
25	-6.9	41.5	811	1005	0.81	150		A. Almeida, 2004 [d]
26	11.6	43.3	730	849	0.86	400		M. Marignani, 2004 [d]
27	-4.0	37.7	474	1167	0.41	83		López-Granados et al., 2004
28	-4.5	37.5	516	1252	0.41	204		Gálvez et al., 2004
I1	-1.4	37.7	318	1129	0.28	154	0.12	Fieldwork 2004
I2	-1.5	38.2	349	1129	0.31	204	0.18	Fieldwork 2004
I3	-1.5	38.2	349	1129	0.31	182	0.05	Fieldwork 2004
I4	-1.5	38.1	349	1129	0.31	151	0.08	Fieldwork 2004
I5	-1.5	38.1	365	1130	0.32	175	0.06	Fieldwork 2004
I6	-1.5	38.2	349	1129	0.31	94	0.18	Fieldwork 2004
I7	-1.5	38.0	365	1130	0.32	123		Fieldwork 2004
I8	-1.0	39.1	502	1073	0.47	205	0.14	Fieldwork 2004
I9	-1.0	39.1	502	1073	0.47	341	0.27	Fieldwork 2004
I10	-1.0	39.0	502	1073	0.47	213	0.22	Fieldwork 2004
I11	-1.0	39.0	502	1074	0.47	275	0.17	Fieldwork 2004

Id	Long. (decimal degrees)	Lat. (decimal degrees)	P [a] (mm)	ET_0 [b] (mm)	HI [c] ($mm\ mm^{-1}$)	Tree density ($trees\ ha^{-1}$)	Projected canopy cover ($m^2\ m^{-2}$)	Source
I12	-0.9	38.9	542	1070	0.51	240	0.16	Fieldwork 2004
I13	-0.9	38.9	542	1071	0.51	177		Fieldwork 2004
I14	-0.8	38.9	478	1071	0.45	171	0.25	Fieldwork 2004
I16	-0.8	38.9	478	1071	0.45	301	0.24	Fieldwork 2004
I17	-0.8	38.8	479	1073	0.45	201	0.07	Fieldwork 2004
I18	-0.9	38.8	479	1073	0.45	198	0.17	Fieldwork 2004
I19	-4.2	37.5	549	1147	0.48	43	0.21	Fieldwork 2004
I20	-4.2	37.5	549	1147	0.48	86	0.23	Fieldwork 2004
I21	-4.0	37.5	522	1147	0.45	136	0.35	Fieldwork 2004
I22	-4.0	37.5	522	1155	0.45	109	0.37	Fieldwork 2004
I23	-4.0	37.5	522	1155	0.45	89	0.32	Fieldwork 2004
I24	-4.0	37.9	434	1217	0.36	87	0.30	Fieldwork 2004
I25	-4.0	37.9	434	1217	0.36	129	0.22	Fieldwork 2004
I26	-4.0	37.8	434	1217	0.36	149	0.32	Fieldwork 2004
I27	-4.0	37.8	434	1218	0.36	76	0.18	Fieldwork 2004
I28	-4.0	37.8	434	1218	0.36	99	0.20	Fieldwork 2004
I29	-4.0	37.8	434	1218	0.36	89	0.31	Fieldwork 2004

Source of climate data: ATEAM database (New et al., 1999; Mitchell et al., 2002)

[a] Annual precipitation

[b] Reference evapotranspiration calculated with the Hargreaves equation (Hargreaves & Samani, 1985; Allen et al., 1998)

[c] Humidity index. $HI = P / ET_0$.
[d] Personal communication with colleagues.