Universitat de Girona

# DEVELOPING A DESERTIFICATION INDICATOR SYSTEM FOR A SMALL MEDITERRANEAN CATCHMENT: A CASE STUDY FROM THE SERRA DE RODES, ALT EMPORDÀ, CATALUNYA, NE SPAIN

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by

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# Developing a desertification indicator system for a small Mediterranean catchment: a case study from the Serra de Rodes, Alt Empordà, Catalunya, NE Spain

**Tesi Doctoral**

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# ABSTRACT

The aim of this research was to develop a simple methodology for precisely appraising the status and trends of desertification in a semi-arid Mediterranean catchment, through a so-called desertification indicator system (DIS).

The assessment of land degradation processes at plot scale was conducted through the monitoring of runoff-erosion microplots. As a result, a set of variables such as soil erosion, soil organic matter and vegetation cover, were identified as the most important factors for soil quality in the target area, and some of these were applied as inputs in the DIS model, accounting for their relevance not only at the plot but also at catchment scale.

Regarding the parameter sensitivity of the DIS model, the saturated hydraulic conductivity as well as the erodability factor were identified as the most sensitive variables, whereas the soil-b parameter and also the vegetation cover and the slope angle were revealed as those affecting soil erosion and overland flow the least. Likewise, the model showed greater sensitivity to the dry than to the normal or wet rainfall scenarios.

From the results of a plot scale model validation exercise it may be concluded that the behaviour of runoff and erosion at plot scale is somewhat different to that at the landscape scale: a scaling problem. At the plot scale, soil erosion was greatly overestimated by the model in the least vegetated environments and especially under cultivated olive trees, whilst it was slightly underestimated in the most vegetated ones (e.g. dense cork trees). The same pattern was found for overland flow, although measured and modelled runoff data were in the same order of magnitude and so errors were smaller than for erosion. Nevertheless, results may be considered significant in terms of which land uses are the most and least potentially degraded and in this way, the model fulfils its objective as a desertification support tool as it identifies the patterns of

change expected, if not the magnitudes. The model would need to be more complex, have better and more input data and a regional scale validation if the magnitudes were to be predicted reliably.

Since soil loss is considered the main indicator of soil erosion processes, according to FAO/UNEP/UNESCO (1979), the original landscape as well as the two developed scenarios, one related to a hypothetical landscape after a wildfire and another to a completely cultivated landscape, may be classified as being under low to moderate land degradation. In comparison to the original scenario, both developed scenarios were revealed to have higher soil erosion and runoff rates, especially the cultivated scenario. Hence, these two scenarios seem not to be a sustainable alternative to land degradation processes in the study area. However, a wide range of alternative scenarios may be developed with the DIS model, based on policies of particular relevance to the region and which help to determine the potential desertification consequences of these policies in this spatially complex landscape.

# RESUM

La desertificació és un problema de degradació de sòls de gran importància en regions àrides, semi-àrides i sub-humides, amb serioses conseqüències ambientals, socials i econòmiques com a resultat de l'impacte d'activitats humanes en combinació amb condicions físiques i medi ambientals desfavorables (UNEP, 1994).

La desertificació tot i no ser un fenòmen recent, ha tingut un ressó molt important tant en l'àmbit científic com polític i tant a nivell nacional com internacional durant els últims anys, la qual cosa ha comportat un gran desenvolupament dels mètodes enfocats al seu estudi, avaluació i predicció (Rubio and Bochet, 1998). No obstant, la desertificació és difícil d'estudiar en la seva totalitat a causa de la complexitat dels processos involucrats i les seves interrelacions.

L'objectiu principal d'aquesta tesi va ser el desenvolupament d'una metodologia simple per tal de poder avaluar de forma precisa l'estat i l'evolució de la desertificació a escala local, a través de la creació d'un model anomenat sistema d'indicators de desertificació (DIS). En aquest mateix context, un dels dos objectius específics d'aquesta recerca es va centrar en l'estudi dels factors més importants de degradació de sòls a escala de parcel.la, comportant un extens treball de camp, analisi de laboratori i la corresponent interpretació i discussió dels resultats obtinguts. El segon objectiu específic es va basar en el desenvolupament i aplicació del DIS.

L'àrea d'estudi seleccionada va ser la conca de la Serra de Rodes, un ambient típic Mediterràni inclòs en el Parc Natural del Cap de Creus, NE Espanya, el qual ha estat progressivament abandonat pels agricultors durant el segle passat. Actualment, els incendis forestals així com el canvi d'ús del sòl i especialment l'abandonament de terres són considerats els problemes ambientals més importants a l'àrea d'estudi (Dunjó et al., 2003).

En primer lloc, es va realitzar l'estudi dels processos i causes de la degradació dels sòls a l'àrea d'interés. En base a aquest coneixement, es va dur a terme la identificació i selecció dels indicadors de desertificació més rellevants. Finalment, els indicadors de desertificació seleccionats a escala de conca, incloent l'erosió del sòl i l'escolament superficial, es van integrar en un model espaial de procés.

El model va ser verificat i validat amb dades de camp obtingudes de la monitorització de varies parcel.les d'erosió instal.lades a la conca. L'anàlisi de sensitivitat del model va permetre determminar els factors clau en base a la seva sensitivitat o insensitivitat a canvis en les dades de camp utilitzades com input en el DIS. Finalment, les dues simulacions del model, representatives de possibles alternatives d'ús del sòl en l'àrea d'estudi, són un exemple the les implicacions potencials de l'aplicació d'aquestes hipotètiques polítiques de desertificació.

Així doncs, l'estudi dels processos de degradació dels sòls a escala de parcel.la es va dur a terme a través del monitoreig de les microparcel.les d'erosió i escolament instal.lades aleatòriament al llarg de dos transectes altitudinals a la conca, abarcant tots els diferents úsos del sòl. Com a resultat, un conjunt de variables com ara l'erosió del sòl, la matèria orgànica i la cobertura vegetal, van ser identificades com els factors més importants de la qualitat del sòl a l'àrea d'estudi, i algunes d'aquestes van ser aplicades com inputs en el DIS, per ser considerades rellevants tant a escala de parcel.la com de conca.

Pel que fa a la sensitivitat del model, la conductivitat hidràulica a saturació i el factor d'erodabilitat del sòl, van ser identificades com les variables més sensitives, mentre que el paràmetre b del sòl i també la cobertura vegetal i la pendent del terreny van resultar ser les que menys influència tenen sobre l'escolament superficial i l'erosió. Així mateix, el model va mostrar més sensitivitat a l'escenari sec que no pas al normal o al plujós.

Dels resultats de la validació del model amb dades obtingudes de l'estudi de les parcel.les es pot concluir que el comportament de l'escolament superficial i l'erosió a escala de parcel.la és força diferent al de a escala de conca, posant de manifest un possible problema d'escala important.

A escala de parcel.la, l'erosió del sòl va ser àmpliament sobrestimada pel model en els ambients menys vegetats i especialment en la olivera cultivada, mentre que va ser subestimada lleugerament en aquells amb més vegetació, com per exemple en les sureres. Aquest mateix comportament va ser observat per l'escolment superficial, tot i que les dades simulades i les observades tenen el mateix ordre de magnitud i per tant els errors han sigut considerats menys importants que en el cas de l'erosió.

No obstant, els resultats poden ser considerats significatius en quant a la identificació dels úsos del sòl més i menys potencialment degradats o en fase de degradació. Per tant, des d'aquest punt de vista el model compleix amb l'objectiu de servir com a eina de suport de desertificació, identificant comportaments de canvi esperats encara que no les magnituds. El DIS necessitaria ser més complexe, tenir millor i més inputs i ser validat a escala regional en el cas de voler ser més precís i acurat.

Ja que el sòl és considerat el principal indicador dels processos d'erosió, segons la FAO/UNEP/UNESCO (1979), tant el paisatge original així com els dos escenaris d'ús del sòl desenvolupats, un centrat en el cas hipotétic del pas d'un incendi forestal, i l'altre un paisatge completament cultivat, poden ser ambients classificats sota baixa o moderada degradació. En comparació amb l'escenari original, els dos escenaris creats van revelar uns valors més elevats d'erosió i escolament superficial, i en particular l'escenari cultivat. Per tant, aquests dos hipotètic escenaris no semblen ser una alternativa sostenible vàlida als processos de degradació que es donen a l'àrea d'estudi. No obstant, un ampli ventall d'escenaris alternatius poden ser desenvolupats amb el DIS, tinguent en compte les polítiques d'especial interés per la regió de manera que puguin contribuir a determinar les conseqüències potencials de desertificació derivades d'aquestes polítiques aplicades en aquest escenari tan complexe espaialment.

En conclusió, el model desenvolupat sembla ser un sistema força acurat per la identificació de riscs presents i futurs, així com per programar efectivament mesures per combatre la desertificació a escala de conca. No obstant, aquesta primera versió del model presenta varies limitacions i la necessitat de realitzar més recerca en cas de voler desenvolupar una versió futura i millor del DIS.

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gemma

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# 1. RESEARCH CONTEXT, AIMS AND OBJECTIVES



## 1.1. THE DESERTIFICATION ISSUE

Historical evidence during the last few centuries shows three main epicentres of desertification: the Mediterranean region, Mesopotamia and the loessial plateau of China, with serious and extensive land deterioration (Dregne, 1976). Thus, desertification is by no means a new phenomenon, despite the attention focused upon it in recent years. Desertification is now worldwide considered as an important problem.

Desertification is a land-degradation problem of major importance in the arid, semiarid and sub-humid regions of the world (Dregne, 1976). Land degradation happens in many different contexts: in cool dry and warm dry environments; in very arid and semi-arid climates; on different types of soil and in very different societies: ancient and modern; advanced technology and traditional; rich and poor; capitalist and socialist and so on.

There is a tendency to blame desertification upon the generated land pressure due to the rapidly expanding population over the middle 20th century, and although it has greatly exacerbated this situation, there is a wider range of other causes to take into account.

### 1.1.1. DESERTIFICATION: SOME DEFINITIONS AND EVOLUTION OF THE CONCEPT

The word *Desertification*, from an etymological point of view, is derived from Latin: on the one hand, "*desert*", with a twofold origin: (1) the adjective *desertus*, that means *uninhabited*, and (2) the noun *desertum*, that means *a desert area*; and on the other hand, "*fication*", that refers to the act of doing (Mainguet, 1999).

Researchers of different disciplines such as climatology, soil science, hydrology, agronomy, geography, political science and economics are involved in understanding the complex phenomenon of desertification. Moreover, a great diversity and confusion among definitions of desertification exists, leading to miscommunication among researchers, among policy-makers and most importantly, between researchers and policy-makers (Glantz and Orlovsky, 1983). According to Glantz (1977), the word desertification has more than 100 definitions, testimony of the complexity of the problem (and the variety of stakeholders to whom that it is relevant).

In general, the common point on which all the definitions agree is that desertification is viewed as an adverse environmental process. The negative descriptors used in these definitions of desertification include: deterioration of ecosystems (Reining, 1978), degradation of various forms of vegetation (Le Houerou, 1975), destruction of biological potential (UNCOD, 1978), decay of a productive ecosystem (Hare, 1977), reduction of productivity (Kassas, 1977), decrease of biological productivity (Kovda, 1980), alteration in the biomass (UN Secretariat, 1977), intensification of desert conditions (Meckelein, 1980; WMO, 1980), and impoverishment of the ecosystem

(Dregne, 1976). Each of these terms suggests change from a favoured or preferred state (with respect to quality, social value, or ecological stability) to a less favoured one.

The definition of desertification has had a progressive evolution over time since the term "desertification", was used for the first time by Aubreville (1949), in his report: "Climats, Forets et Desertification de l'Afrique Tropicale". The first meaning related to the desertification concept, took into account the desert areas as the starting points from where desertification begins, with the consequence of desert encroachment. Afterwards, the Desertification Map of the World (UNEP/ FAO/UNESCO/WMO, 1974), adopted a broad definition of the concept*:*

*"Desertification is the intensification or extension of desert conditions. It is a process leading to reduced biological productivity with consequent reduction in plant biomass as in the lands carrying capacity for livestock, in crop yields and human well being"*

The United Nations Conference on Desertification in Nairobi in 1977 (UNCOD, 1978), for the first time, considered desertification as a major environmental problem at a global scale. The Conference served to draw attention to the desertification problem and it addressed this phenomenon as a process of reduction of the biological potential of the soil capable of leading to desert conditions, thereby basing the concept on productivity criteria and out of consideration of the geographic positions (polar or tropical) of interested areas, climatic characteristics, causes (natural or human) and processes (i.e. salinization and acidification).

The next meaning of the concept desertification, was written by Garduno (1977):

*"Desertification is the impoverishment of arid, semi-arid, and some subhumid ecosystems by the impact of human activities. I is the process of change in these ecosystems that leads to reduced productivity of desirable plants, alterations in the biomass and diversity of life forms, accelerated soil degradation, and increased hazards for human occupancy."*

Mabbutt (1978) wrote more about the same above quoted idea:

*"…the change in the character of land to a more desertic condition involving the impoverishment of ecosystems as evidenced in reduced biological productivity and accelerated deterioration of soils and in an associated impoverishment of dependent human livelihood systems"*.

Kates et al. (1977) and later Street (1987), defined the desertification process as and intricate process of land degradation.

*"It involves destructive processes in which the productive base deteriorates and the social system is imperilled. Unlike drought, which is usually a short-term diminution of available moisture, the physical processes involved in desertification are long-term, chronic, and pervasive."*

Wehmeier (1980), pointed that the term of desertification should comprise some notion of long-term and possible irreversible or irreparable change:

*"Desertification here will be understood as disadvantageous alterations, not oscillations, of and within ecosystems in arid and semiarid areas. These alterations are usually triggered by man (involuntarily), take place within a relatively short period of time (several years to several decades), and very often cause irreparable or but partly repairable damage"*.

Some authors proposed a definition of desertification emphasizing the role of human activities and related impacts. Accordingly, Dregne (1982) defined the concept as:

*"The impoverishment of terrestrial ecosystems under the impact of man. It is the process of deterioration in these ecosystems that can be measured by reduced productivity of desirable plants, undesirable alterations in the biomass and the diversity*

*of the micro and macro fauna and flora, accelerated soil deterioration, and increased hazards for human occupancy",*

Sabadell et al. (1982) pointed the possible importance of climatic controls but gave them a relatively inferior role:

*"Desertification is the sustained decline and/or destruction of the biological productivity or arid and semiarid lands caused by man-made stresses sometimes in conjunction with natural extreme events. Such stresses if continued or unchecked over a long period lead to ecological degradation and ultimately to desert like conditions".*

The UNEP/ FAO (1984), defined the desertification process as a problem more from a human perspective, including the general contributory factors, the areas affected and the general environmental outcome:

*"…a comprehensive expression of economic and social process as well as those natural and induced ones which destroy the equilibrium of soil, vegetation, air and water, in the areas subject to edaphic and/or climatic aridity".*

Hare (1985), introduced the idea of natural irreversible deterioration, thus, the meaning of the concept desertification is related to: *"The ultimate non productive and desert-like stage of deteriorated environment"*, desertification is the name given to the processes whereby such ecosystems lose this capacity to revive or to repair themselves.

In the same direction, a more recent definition by UNEP/UN (1990), considered the process of desertification as: *"…land degradation in arid, semi arid and dry subhumid areas resulting mainly from adverse human impact"*; the ultimate stage of land degradation, the point when land becomes irreversibly sterile in human terms and with respect to reasonable economic limitations. This latter definition was only partially accepted at the Soil Conference in Rio de Janeiro in 1992, which suggested the following revision of the definition:

*"Desertification is land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors including climatic variations and human activities".*

Mainguet (1994), after a large review of the different definitions of the desertification process, considered the proposed by the UNCOD (1978), the most comprehensive, precise and accurate one, and concluded defining the process as:

*"...desertification is best reserved for the ultimate step of land degradation, the point when land becomes irreversibly sterile in human terms and with respect to reasonable economic limitations".*

The latter definition is internationally accepted as the most complete for describing the desertification phenomenon, being the one officially considered by the UNCOD (1978). Hence, this project will also employ this definition as the theoretical and conceptual basis for the development of the desertification indicator system.

#### 1.1.1.1. Desertification versus land degradation and soil degradation

##### 1.1.1.1.1. Land Degradation

Land degradation describes how one or more of the land resources such as soil, water, vegetation, rocks, air, climate and relief, has changed for the worse (Stocking and Murnaghan, 2001). The change may prevail only over the short term, with the degraded resource recovering quickly. Alternatively it may be the precursor of a strong deterioration process, causing a long-term, permanent change in the status of the resource. It therefore includes changes to soil quality, the reduction in available water, the diminution of vegetation sources and of biological diversity, and the many other ways in which the overall integrity of land is challenged by inappropriate use.

#### 1.1.1.1.2. Soil Degradation

Soil is defined as a natural, three-dimensional body with definable boundaries that commonly, but not always, consists of horizons made up of mineral and organic materials, contains living matter, and can support vegetation (Soil Survey Staff, 1996). Soil degradation is defined by FAO/ UNEP/ UNESCO (1979) as: "a process which lowers the current and/ or the potential capability of soil to produce (quantitatively and/ or qualitatively) goods or services. Soil degradation is not necessarily continuous. It may take place over a relatively short period between two states of ecological equilibrium". The processes of soil degradation are mainly water erosion, wind erosion, salinization and/ or sodification, chemical degradation, physical degradation and biological degradation. Soil degradation is considered as the most critical component of land degradation and, in the framework of irreversible land degradation, as the main factor of desertification (Mainguet, 1994).

#### 1.1.1.1.3. Desertification versus land degradation

Although both terms "*desertification*" and "*degradation*" end up with partially or totally unproductive soil, they are not synonymous (Balba, 1995). On the one hand, degradation is concerned with changes in the soil physical, chemical, and biological properties, which affect the soil as a medium for plant growth (FAO/ UNEP/ UNESCO, 1979). On the other hand, desertification pays more attention to environmental factors, which affect the soil productivity. Desertification takes place gradually and under variable conditions and its processes may also vary, but the result is always the same: the change of productive to unproductive land and the expansion of the desert area (Balba, 1995). Both terms are affected significantly by human activities.

This project will focus on the study of the desertification problem in the Mediterranean region, although some parts of the study such as the literature review of the causes, processes and indicators will address both concepts, land degradation as the general process and desertification as more specific phenomenon, since they are closely interrelated.

#### 1.1.1.2. Desertification in a Mediterranean context

In a Mediterranean context, desertification is defined as the consequence of a set of important processes, which are active in arid and semi-arid environments (Kosmas et al., 1999). Desertification and the consequent degradation of vulnerable ecosystems are problems that occur in all the European countries of the northern Mediterranean. These Mediterranean countries such as: Spain, Portugal, Italy and Greece, are considered by the UNCOD as potentially degraded areas. This is because these areas share some or all of a number of common features and problems identified by the UNCOD in its Annex IV: (a) "Semi-arid climatic conditions affecting large areas, seasonal droughts, very high rainfall variability and sudden and high intensity rainfall; (b) poor and highly erodible soils, prone to develop surface crusts; (c) uneven relief with steep slopes and very diversified landscapes; (d) extensive forest coverage losses due to frequent wildfires; (e) crisis conditions in traditional agriculture with associated land abandonment and deterioration of soil and water conservation structures; (f) unsustainable exploitation of water resources leading to serious environmental damage, including, …, salinization and exhaustion of aquifers; and (g) concentration of economic activity in coastal areas as a result of urban growth, …, tourism and irrigated agriculture".

According to the DESERTLINKS project (2001), in general terms, desertification in the European Mediterranean countries is linked to: (a) "The particular climatic and geomorphological characteristics which, combined with often poorly adapted use of land, results in a highly vulnerable environment", (b) "Strong human pressure from agricultural and pastoral activities for at least four thousand years, often in conjunction with phases of demographic growth and rapid economic development", and (c) "An enormous increase in the human pressure recorded after the 1950s, accompanied by major economic transformations such as mechanisation and intensification of agro-pastoral practices, and a strong increase in water demand linked to urban and tourism development".

As in most other semi-arid regions, desertification in the Mediterranean region is largely a human-driven problem, which can be effectively managed only through a thorough understanding of the principal ecological, socio-cultural and economic driving forces associated with land use and climate change, and their impacts (MedAction, 2001).

**1.1.1.2.1. Desertification in Spain**

All of the arid regions of Spain have been moderately to severely desertified for decades, if not for centuries or even millennia. Approximately 50 percent of Spain is arid. In the arid regions about 70 percent is moderately desertified and 30 percent is severely desertified (Dregne, 1986).

Heavy grazing and woodcutting have done most of the damage, but wind and water erosion on cultivated land has also been extensive. During the past several centuries, heavy grazing by sheep and goats has led to the destruction of much of the herbaceous and woody vegetation on the non cultivated land (Albareda, 1955). Water erosion has been severe on the overgrazed slopes as a result of the loss of vegetative cover and the torrential character of rains. A monoculture of grain in the cultivated regions has depleted the original fertility of the soil and has been responsible for increasing the susceptibility of the land to wind and water erosion. Extended droughts from time to time have served to accelerated desertification (Puigdefàbregas, 1998). Water erosion is severe nearly everywhere on sloping land (Thornes, 1995). Salinization and waterlogging of irrigated land is a major problem in some parts of the irrigated valleys in north-eastern of Spain. Seepage water from irrigation on the higher land has caused waterlogging and salinization of lower-lying areas (Martínez Beltran, 1978). Destruction of the native vegetation and the subsequent water erosion of thin soils on slopes has had a devastating effect on the plant environment and on the potential productivity of the land (Dregne, 1986).

## 1.2. SOCIO-ECONOMIC AND ENVIRONMENTAL FRAMEWORK

### 1.2.1. INTERNATIONAL PERSPECTIVES OF DESERTIFICATION

It is well documented that land degradation constitutes a major worldwide problem with dramatic environmental, social and economic consequences (see Dregne, 1976). According to the United Nations Convention to Combat Desertification (UNCCD), today over 250 million people are directly affected by desertification and some one billion in over one hundred countries, are at high risk (Lean, 1995). Growing awareness in the international community of the global scale of land-degradation problems and the need for global responses has led to an increasing number of international initiatives in understanding, adaptation and mitigation.

Desertification began to emerge as a major global environmental issue in the early 1970s, when a severe drought lead to the death of over 200,000 people and millions of animals in Africa. In 1977, the United Nations Conference on Desertification (UNCOD) held in Kenya, addressed desertification as a worldwide problem for the first time. A Plan of Action to Combat Desertification (PACD) was adopted from the conference, although because of scarce resources its implementation was limited. The subsequent action undertaken at an international level was the United Nations Conference on Environment and Development (UNCED) organized in Rio de Janeiro, Brazil, in 1992. The conference, also known as the Earth Summit, highlighted the problem of desertification and its importance in impeding sustainable development and encouraged the establishment of an International Negotiating Committee (INCD) to prepare a Convention to Combat Desertification in those countries affected by drought and/or desertification. In 1994 the United Nations Convention to Combat Desertification (UNCCD) was adopted in Paris, France. The aim of this treaty is to prevent and reduce land degradation, rehabilitate partly degraded land and reclaim desertified land. It deals with the extensive desertification phenomena of the northern Mediterranean countries and also addresses the problem in Central and Eastern Europe. The elaboration and implementation of Regional Action Programmes (RAPs) and National Action Programmes (NAPs), as foreseen by the UNCCD, form valuable policy instruments to combat desertification and soil degradation phenomena in these areas. To date over 170 countries have acceded or ratified the Convention clearly demonstrating the global nature of the problem of desertification (Lean, 1995).

In this same context, other important actions directly or indirectly undertaking land degradation problems have been conducted recently: (a) the *Framework Convention on Climate Change* FCCC (1992), which recognises the role and importance of terrestrial ecosystems as sinks of greenhouse gases and that land degradation problems and changes in land use can exacerbate the emission of gases to the atmosphere; (b) the *Convention on Biological Diversity* CBD (1992) which aims to conserve biological diversity and encourage the sustainable use of its components, with fundamental concerns that biological diversity is being significantly reduced by human activities, including soil and land degradation; (c) the *Kyoto Protocol* (1997) which promotes sustainable development; (d) the *European Community Biodiversity Strategy* (1998) and in particular, its Action Plan on the Conservation of Natural Resources which includes an action establishing an information base on soil erosion; (e) the *European Soil Forum* ESF (1999), which was created with the role of providing a better understanding of soil protection issues and to promote the exchange of information among participating countries; and (f) the *World Summit on Sustainable Development* (WSSD) held in Johannesburg, South Africa, in 2002. This latter Summit evaluated the obstacles to progress and the results achieved since the 1992 Earth Summit, in order to provide a new impetus for commitments of resources and specification towards global sustainability (Gardiner, 2002).

### 1.2.2. EUROPEAN PERSPECTIVES OF DESERTIFICATION

Desertification and the consequent degradation of vulnerable ecosystems are problems that occur in all the European countries of the northern Mediterranean. It is specifically recognised, by the European Union member states themselves and by UNCCD, as occurring in areas of Portugal, Spain, Italy and Greece. The European Commission and its northern Mediterranean member states are signatory partners of the UNCCD.

For the period 2003-2006, the European Commission through the Sixth Framework Programme (FP6), promotes the development of activities with the aim of integrating and strengthening European research. One of the five main Activity Areas is: “Sustainable development, global change and ecosystems”, which through the action: “Global Change and Ecosystems”, addresses the specific priority area of “Mechanisms of Desertification”. This section directly concerns the most relevant topic of the present study, aiming to: “study the driving processes of desertification in the framework of likely scenarios of multiple stresses driven by land use changes and climate change and the development of methods and tools to achieve an integrated assessment”. Special emphasis is made on the identification of a threshold-indicators framework, the assessment of the effect of extreme conditions on erosion processes and land degradation and the development of advanced modelling tools and actions. In addition, it also established that the above-mentioned strategies have to be developed in areas relevant to the UN Convention to Combat Desertification, such as Spain, studying which the study site is located. Regarding land degradation and desertification studies, it is well known that soil erosion and particularly water erosion is the most important process in the Mediterranean region (Poesen, 1995). In this same context, the actions taken under the sub-priority “Water cycle, including soil-related aspects” are also of important relevance for this project. Common features are taken into account in both the sub-priority and the research project addressing soil-water processes, the influence of different climatic scenarios and vegetation cover as well as land use changes on the development of a soil-water spatial and numerical models for the assessment of desertification at a local scale. With regard to: "The Soil Protection Communication" (DG ENV 2001) of the European Commission, this report based on the 6th Environment Action Programme, "Our Future, Our Choice" (COM (2001) 31 final) and the Sustainable Development Strategy Communication (COM (2001) 264 final), which highlights the need for a strategy of soil protection, including soil erosion, desertification, land degradation, and hydro-geological risks, specially in mountain and arid areas. These actions are also directly or indirectly integrated in the objectives of the present study, contributing to a better understanding of these important issues at a local level.

Although explicit Community-level policy focused on soil protection does not exist, due to the multifunctional role of soil and its universal presence, a broad range of Community instruments influence soil protection and for instance land degradation and desertification. The most prominent policies are listed as follows:

(a) The *Common Agricultural Policy* (CAP), which takes into account the protection of the soil, considering it as the most important factor for agricultural production.

(b) *Regional Policy: Structural and Cohesion Funds*, which support programmes such as LEADER, INTERREG and URBAN based on the improvement and protection of the soil, for instance including investments to combat erosion, to plants forests, to prevent flooding, and to rehabilitate derelict and polluted land, as do support for controlled tourism and leisure activities.

(c) *Environmental Policy:* The close link between soil and the other major compartments such as water and air is taken into consideration in specific environmental legislation. Soil protection is one of the aims of this policy.

(d) *Research Policy*: In the context of various Community research programmes, a number of soil-protection problems are addressed. For example, the PESERA project works on soil-erosion risk assessment all over Europe. The 5th Research Framework programmes "Environment and Sustainable Development" and "Quality of Life" also supported soil-related research. The European Soil Bureau, a specific project of the Commission's Joint Research Centre, is a network of soil-science institutions, which is carrying out scientific and technical work programmes in order to collect, harmonise and distribute soil information from countries all over Europe relevant to Community and national policies.

Moreover, specific desertification research efforts at the European scale have also been promoted by the European Commission including projects such as: EU EFEDA, MEDALUS, MODMED, MODULUS, MEDACTION and DESERTLINKS, which have involved researchers from universities and research centres of different European countries working together using an interdisciplinary approach in order to take into account all aspects concerning desertification. These projects have focused their efforts on providing reliable data and information sources, to underscore the understanding of the causes of desertification, in order to forecast and combat future desertification, as well as to mitigate the effects of on-going processes. The study has been carried out at both, local and regional scale, and has taken into account environmental and socio-economical factors characteristics of the Mediterranean area.

### 1.2.3. SPANISH PERSPECTIVES ON DESERTIFICATION

Spain as a Mediterranean country and a Member State of the European Union, has taken specific initiatives and actions on soil protection, which tend to concentrate on erosion and desertification, mainly within the context of the UN Convention to Combat Desertification (UNCCD).

The application of the UNCCD in Spain is integrated in the National Environmental Policy through the Environmental Ministry, and its short-term objectives are mainly focused on the rational use and recuperation of the water quality and its environment, the monitoring of the loss of vegetation cover (deforestation, wildfires) and the production of the soil (erosion and desertification), the recuperation of the coastal littoral, the reduction and management of wastes, as well as the conservation and recuperation of the natural heritage (Ministerio de Medio Ambiente, 2000). Specifically, with reference to the present study it has to be noted that one of the key actions set up by the Ministry of Environment is the creation of a working group with the aim of defining and applying indicators of desertification. These indicators must be integrated in the National System of Environmental Indicators framework using the DPSIR model, which consists of the study of the different types of indicators: driving-forces, state, pressure, response and impact. Several organisations such as the OCDE, the Sustainable Development Commission of the United Nations and the European Environmental Agency have adopted this system (Ministerio de Medio Ambiente, 2000).

In Spain, the National Programme to Combat Desertification was initiated in 2000, with an assessment of the seriousness of desertification, which identified and enumerated watersheds under several grades of desertification risk. It concluded that 31 % of the land area of the country is under serious threat of desertification. Policies, which are affecting desertification, have been identified and a set of specific actions has been designed: follow up will include a set up of relevant indicators and the upgrade of the network of desertification experimental stations. A set of generic actions regarding prevention and restoration of desertified areas, sustainable management of water resources and forest fire prevention have been planned. Finally the programme assesses the impact on desertification of current EU policies and their proposed revision.

## 1.3. AIM AND SPECIFIC OBJECTIVES

### 1.3.1. AIM

The aim of this study is to develop a desertification indicator system (DIS) for a semi-arid Mediterranean catchment. This system will allow the formalization of desertification indicators through process-based models. The DIS will also allow the identification of present and future threats to allow the development of effective measures to combat desertification. It will establish an information system to monitor the bio-physical and chemical factors, as well as assess the extent, intensity and severity of land degradation processes in the target area currently and for a series of policy scenarios. The system will thus focus on the better understanding of the driving factors, which lead to desertification in the chosen catchment, and the factors likely to exacerbate the issue through the unforeseen implications of local-scale policy decisions.

### 1.3.2. SPECIFIC OBJECTIVES

To achieve the aim, effort is focused on two principal approaches: on the one hand the assessment of the most important factors of land degradation at the plot scale and on the other hand the development and application of a mathematical model at the landscape scale.

#### 1.3.2.1. Assessment of the most important factors of land degradation at plot scale

The assessment of the most important factors of land degradation at plot scale, will mainly entail fieldwork determinations, laboratory analyses and their correspondent interpretation and assessment, which will involve several objectives:

1. To better understand the global processes and causes of land degradation, as well as, the specific ones for the target area.

2. To evaluate the role of land-use/cover on the main soil physico-chemical properties at the plot scale.

3. To assess land-degradation processes on the basis of the monitoring of runoff-erosion microplots installed in different land uses in the target area for a series of scenarios of events over a full hydrological year.

4. To develop soil-quality indices relevant at the plot scale.

#### 1.3.2.2. Development and application of the DIS model

The development and application of the DIS model will involve GIS techniques in combination with modelling work, which will be carried out in order to see which of the most important factors at the plot scale will remain important at the landscape scale and which factors which are not important at the plot scale will become important at the landscape scale. The development and application of the DIS will entail several phases:

1. To identify the most relevant indicators regarding land degradation at a local to regional scale, and specifically the ones important in the study area, on the basis of the undertaken literature review on the processes and causes of land degradation and desertification.

2. To develop a set of potential indicators of desertification, on the basis of this literature review, in order that these can be applied in the development of the DIS. From this set, a subset relevant to the study area and study problem will be derived.

3. To develop spatial datasets for the study region on the basis of existing cartographic data and from which some of the indicators and model parameters can be derived in order to develop spatial indicators of desertification process and risk.

4. To collect relevant meteorological data in order to parameterise these models spatially for the study region.

5. To build and verify a process-based hydrological and soil-erosion model based on well-established representation of the processes and to link these submodels together within the context of a high spatial and temporal resolution model run within a GIS.

6. To carry out a sensitivity analysis of the verified model under dry, normal and wet conditions in order to ensure parsimony of the model and to understand better the sensitive and thus important physical parameters for desertification in this environment.

7. To validate/test the model at the plot scale by comparison with particular events over particular examples of the microplots discussed under the section 1.3.2.1., of the specific objectives.

8. To develop scenarios for land use change based on policies of particular relevance to the region and to use these scenarios as inputs to DIS simulations in order to determine the potential desertification consequences of these policies in a spatially complex landscape.

## 1.4. STUDY AREA

### 1.4.1. THE SERRA DE RODES CATCHMENT AS A PART OF THE NATURAL PARK OF CAP DE CREUS

#### 1.4.1.1. The Natural Park of Cap de Creus

The peninsula of Cap de Creus is the eastern part of the Iberian Peninsula. It is located in the Alt Emporda region and occupies an area of 142 $km^2$. This zone is considered a space of first order, of significant biological diversity and is an impressive geological and biological feature, with unique examples of maritime and littoral terrestrial environments (Franquesa, 1995).

The Natural Park of Cap de Creus was officially created in 1998, becoming the first maritime and terrestrial Natural Park of the Iberian Peninsula. The Park comprises a total area of 13,886 hectares, of which 10,813 correspond to the terrestrial and 3,073 to the maritime environment (Franquesa, 1995). Eight municipalities are located within the Park: Cadaqués, Port de la Selva, Selva de Mar, Llançà, Palau-Saverdera, Roses and Vilajuïga.

By law, in the Natural Park three different protection levels are established (in increasing order of protection): natural park zones, natural sites of national interest and natural reserves. Protection and preservation measures specifically for the terrestrial environments are related to the geological, botanical, faunal and landscape values, as well as the different elements of cultural interests in the area, and are legally punished in case of negligence.

Therefore, there exist multiple reasons that justify the need for protection and conservation of this unique place. In general the protection law of the Natural Park, guarantees the sustainable development of this particular area.

#### 1.4.1.2. The Serra de Rodes area

The study area is part of the Serra de Rodes, in the Natural Park of Cap de Creus. It is listed as a “Natural Site of National Interest”, implying specific limitations and protective actions addressing the environmental management of the territory.

The Serra de Rodes is mostly constituted of steep slopes. For grazing and agricultural purposes, this ecosystem was almost completely terraced in the past leading to the whole transformation of the landscape. The transformation of the vegetation cover and the fluctuations of the land use have a resultant effect on the hydrology and geomorphology, directly interfering on the infiltration rates, the water storage, evapotranspiration, runoff generation and sediment yield as well as nutrient losses.

### 1.4.2. GEOGRAPHICAL LOCATION

The Serra de Rodes catchment, located in the municipality of Vilajuiga, in the Alt Empordà area, Province of Girona, NE Spain (42º 20’ 4’’N, 3º 6’ 19’’E) (Figure1.1.). This Mediterranean site is enclosed in the Natural Park of Cap de Creus and occupies an area of 30 $km^2$, ranging from 60 m to 300 m above sea level (a.s.l.). It is bound by the Mediterranean Sea in the North, by the Roses bay in the East, by the Albera Mountains in the West and by the Empordà plain in the South.

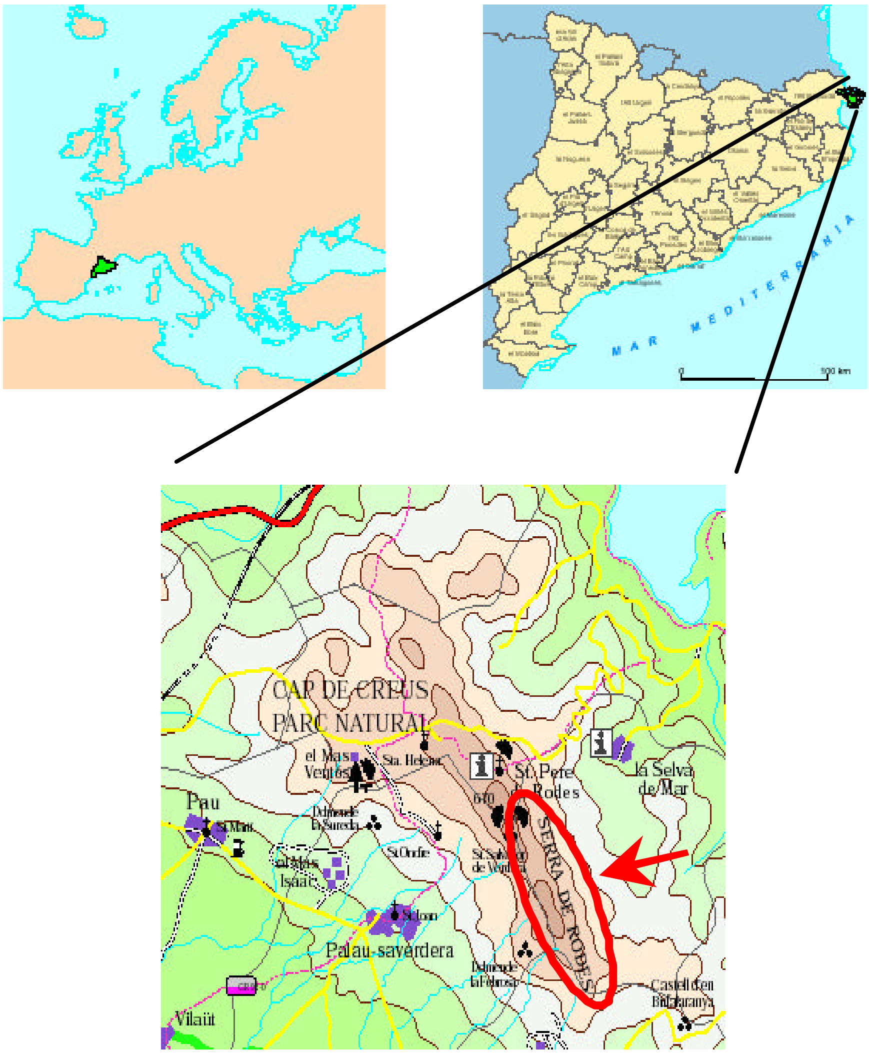


Figure1.1. Location of the study area: Europe-Spain-Catalonia; Catalonia-Alt Empordà; Natural Park of Cap de Creus- Serra Rodes catchment

### 1.4.3. GEOMORPHOLOGY

#### 1.4.3.1. Geology

The Serra de Rodes catchment and in particular the peninsula of Cap de Creus, represents the eastern end of the most axial part of the Pyrenees mountains, originally in a lower position and constituted by materials from the Hercynian period, which later on became the highest part of this mountain range.

This region is characterized by two different lithological units. On the one hand, a sequence of sedimentary rocks in the South-east part, and on the other hand, parent material composed of Palaeozoic acid-ignious rocks in the study area. Thus, the catchment is mainly constituted by granodiorite and granites with albite, orthoclase and microcline (Història Natural dels Països Catalans, 1992). Both units form the two intrusive parts of Roses and Rodes, located in the materials of the high zone of the sedimentary sequence.

From a structural point of view, all of these Hercynian terrains are characterized by multiple periods of tectonic activity, with two main deformation phases. The first one includes the genesis of all the structures formed before the metamorphic climax, and it is characterized by the formation of folds with well developed foliation of the axial plain. The dominant faults formed along a NW-SE axis, control the alienation of the structures at a cartographic level. The mega structure of the surfaces forming the peninsula of Cap de Creus is mainly controlled by the later folding event (Franquesa, 1995).

The current morphology is the result of a recent process conditioned by the interaction between the geologic substratum and the atmosphere, the hydrosphere and the biosphere. The geomorphology of the area has been strongly affected by the *Tramuntana* wind, as well as the salinity of the littoral zones, entailing a landscape composed by peculiar forms due to the cumulative erosion processes over the years.

#### 1.4.3.2. Relief

The peninsula of Cap de Creus, represents the continuation of the Albera mountain range and it is constituted by the Serra de Rodes, located between the Roses bay and the

Port de la Selva village, where the maximum altitude is around 670 metres (at the Sant Salvador summit).

Mountain streams are present in the area in great number, although they have very low flow and are often ephemeral. Despite the fact that the altitude is not very high, the relief is quite steep, due to the action of water erosion processes. Most of the surface has slopes greater than 35º and some times even more than 60º.

### 1.4.4. CLIMATE

In the study area, the climate is believed to play an important role in the genesis of the landscape.

#### 1.4.4.1. General climatic features of the Serra de Rodes catchment

The climate of the Mediterranean biome is classified as xerothermic. This temperate climate is particularly cold in winter and has mild-hot summers. During three or four months in winter the daily average temperatures are very low, although they rarely reach minimums of –5ºC. On the other hand, a very dry month in summer is common. Rainfall events are quite scarce with events usually take place in the autumn and winter season. The annual average rainfall is 550-600 mm and has a high interannual variability.

The climate directly influences the soil moisture and temperature. Factors such as: the textural class, soil organic matter content, and the aeration state, strongly depend on the climate characteristics.

In this area, the climate is severely affected by the effects of the *Tramuntana* wind. This wind, which comes from the Pyrenees, is normally very strong and cold. It is considered a bioclimatic factor of maximum importance in this area. The *Tramuntana* involves a lower humidity level, and consequently the environment becomes dryer and the availability of water in the area decreases.

#### 1.4.4.2. Precipitation

The Mediterranean climate is characterized by a high variation in the amount of rainfall from year to year, although the mean annual values are quite similar. Thus, the total annual precipitation is more than 1000 mm some years and less than 300 mm others.

Monthly rainfall is also seasonally irregular (Franquesa, 1995). October is the wettest month and July the driest. The main rainfall events in terms of amount of water occur between September and December. March is also quite rainy, whereas the rest of spring months are not especially wet. The driest months are the summer season, when rainfall is rare and torrential in nature (Franquesa, 1995).

The irregular precipitation pattern and its distribution over the year, determine the type of vegetation growing in this area, which has to adapt to the scarcity of water during some periods of the year.

#### 1.4.4.3. Temperature

The temperatures in the study area show a typical Mediterranean seasonal oscillation. There are great thermal differences between summer and winter, and so between years. In this mountainous environment, the hottest month is July and the coldest January, with a mean monthly maximum and minimum temperature of 36ºC and 4ºC respectively. In general, the mean annual temperature is set between the isotherms of 15ºC and 16ºC for the peninsula of Cap de Creus.

#### 1.4.4.4. Wind: the *Tramuntana*

The *Tramuntana* is a northerly wind, dry and generally cold and violent, which blows at great velocity some times reaching more than 120 km/hr. It can blow uninterrupted for several days and it is common all over the year. This wind is considered an important factor, for its crucial role on the distribution and form of the vegetation as well as for its positive influence on the evapotranspiration rates.

### 1.4.5. VEGETATION

The current dominant vegetation composing the landscape of the peninsula of Cap de Creus, is linked to the recolonization of cultivated and abandoned lands, as well as grazing activities and the repeated utilization of fire by shepherds in order to produce better pasture.

Figure 1.2. Overview of the terraced landscape mainly covered by scrubland, of the Serra de Rodes catchment.

The vegetation of Cap de Creus, contrary to its rather monotonous appearance at first sight, is very species rich. The flora includes endemic and extremely rare species. For example there are more than thirty different species of fern, which for the Mediterranean territory, represents a large number.

The Natural Park of Cap de Creus and the Serra de Rodes catchment is mainly dominated by shrub communities (Figure 1.2.). Forests occupy less than 2% of the total surface, whereas shrub vegetation represents more than 60% of the cover, excluding the littoral area (Font, 2000).

Some of the most representative shrub communities, and typical for the Mediterranean basin, are the Spanish lavender (*Lavandula stoechas*), an evergreen bush highly resistant to heat and water shortage. Lavender represents the initial stage of the vegetation colonization of the abandoned agricultural dry lands. Scrub of rockrose and bush (*Cisto-Sarothamnetum catalaunici*), are taller and characterized by the dominant presence of the black rockrose (*Cistus monspeliensis*), the white rockrose (*Cistus albidus*) and some leguminous plants such as furze (*Calicotome spinosa; Ulex parviflorus*), a spiny evergreen shrub. When the areas have not been cultivated or have been slightly affected by fires and grazing, species such as heather (*Erica arborea*) grow.

At higher zones and the piedmont a particular type of tall and dense heather (*Lavandulo-Ericetum scopariae; Cytiso-Ericetum arboral*) can be found, as well as *Erico-Arbutetum* species that incorporate plants more demanding in terms of moisture conditions such as the *Erica scoparia*, *Pistacia lentiscus*, *Phillyrea angustifolia, Juniperus oxycedrus, Abutus unedo, Cytisus villosus,* or the *Myrtus communis*.

Apart from shrubs, there are no other dominant elements in the landscape, although sometimes these are mixed with small patches of Mediterranean pasture where the presence of a grass stage constituted by species such as *Brachypodium retusum* and

*Helianthemion guttati* is very common. In some limited areas of the Serra de Rodes, *Quercetum ilicis galloprovinciale* sometimes mixed with cork trees (*Quercus suber),* as well as hazel plantations (*Polysticho-Coryletum*) are also present (Història Natural dels Països Catalans, 1992).

### 1.4.6. SOILS

The study of the vegetation is closely linked to the study of the soil. Both the vegetation and soil are dynamic factors, which evolve influencing each other. Hence, it is very important to know the nature of the substratum of the soil as well as its constitution in order to determine the type of vegetation cover likely in a specific area.

With reference to the vegetation, the fundamental aspects to be considered are the age of the soil and the group of physico-chemical characteristics of the different horizons, especially the depth, the textural class, the structure and the fertility. These parameters condition the water-holding capacity and consequently the ability to support the vegetation community.

Soils in this region are from various lithologies, due to the heterogeneity of the geology and catena effects. Therefore, several types of soils are present in the area.

The Serra de Rodes is an area of very steep slopes. The soil is absent in some parts of the surfaces of the slopes, and it mostly remains in the valleys. Hence, this territory is characterized by shallow and poor developed soils, classified as entisoils (lithic xerorthent) or inceptisoils (distrocrept) according to the Soil Taxonomy System, with the horizon development: A, C/R and a B cambric horizon in the areas of deposition. The erosion conditioned by the topography, is a determinant factor of the depth and soil characteristics of this area.

### 1.4.7. LAND USE

#### 1.4.7.1. Land use and land-use change in the Mediterranean study catchment

Every landscape is the result of the action of several cultures and different socio-economical and demographical conditions in a particular site (Wainwright, 1994). Commonly, this cultural superposition explains the main organization points of a landscape, transformed by human activities in relation to specific objectives, which are never determined by environmental factors.

Nowadays, in developed countries, abandoned fields are one of the key landscape elements of mountain regions. In particular, the abandonment of large, sloping, cultivated areas and the intensification of the best lands, constitute the most relevant characteristic of the evolution of the land uses in most parts of the Mediterranean basin (Ruíz-Flaño,1993).

Over centuries, mountainous areas have been under high pressure. The demographic increase during the 18$^{th}$ and 19$^{th}$ centuries, and the low industrialization of the urban centres, lead to the exploitation of marginal lands. These lands, characterized by steep slopes, rocky and infertile soils, were topographically quite inaccessible, and were located in remote sites far from the cities and villages (Lasanta, 1989).

During the 20$^{th}$ century, an important process of industrialization and a great revitalization of the tertiary sector took place, creating the depopulation of mountain areas, intense migration towards the cities and progressive abandonment of the cultivated fields with steep slopes.

Agriculture of subsistence was practised in this environment over centuries. Most of the ancient soils were cultivated in terraces, with their own hydrologic and geomorphologic dynamics. After their abandonment, these soils evolved naturally experiencing different levels of vegetal colonization and conservation, leading to a landscape characterized by a great diversity and heterogeneity and responding to the different stages of the natural succession. As a result, there exist abandoned fields which are sites very deteriorated, with sparse shrub cover, rocky soils and an intense erosional activity, abandoned with a dense shrub canopy.

The land-use changes experienced in these areas have lead to changes in the hydrological cycle, causing alterations in the dynamics of the nutrients and the sediment yields, in addition to the increase of the vulnerability of these Mediterranean ecosystems through physical perturbations such as wildfires, droughts, or floods.

### 1.4.7.2. Evolution and transformation of the vegetation cover

The peninsula of Cap de Creus is characterized by exceptional climatic and orographic conditions, which together with its peninsular geographical isolation, lead to several interesting types of vegetation. This landscape has been deeply modified due to successive changes and perturbations mainly driven by human activities. Therefore, the landscape is the result of geographic, climatic, edaphic and anthropic factors, which have contributed to its current state.

#### 1.4.7.2.1. Human influence on the landscape development

The current landscape of the Peninsula of Cap de Creus is the result of the strong anthropic influence in the area. The current vegetation communities are the result of the action of humans over centuries and although some forested areas exist, shrubs and meadows dominate the landscape of this site.

Agriculture (mainly the cultivation of vineyards and olive trees), has been a determining factor in the transformation of the landscape. This zone has had two periods of high agricultural development, coinciding with important periods of demographic increase:

one between the 10th and 13th centuries, and the other one during the 18th and 19th centuries, when the agricultural land was extraordinarily extended even in the steepest and roughest parts of the terrain.

#### 1.4.7.2.2. The importance of agricultural practices

In order to obtain new agricultural land, initially the process of deforestation was carried out, followed by the total elimination of the pre-existent vegetation from its root. At the same time, in this terrain with a very complex topography, the construction of terraces was required in order to grade the steep slopes of the mountains with infinite *drystone walls*. This practice allowed the cultivation of these steep and inaccessible areas, severely affecting and modifying the edaphic and hydrological conditions of the environments. Therefore, these practices not only completely transformed the landscape but also modified the infiltration rates, the resistance of the soils to erosion processes and consequently the modification of runoff rates and sediment yields (García-Ruíz et al, 1991).

The dimensions of *drystone walls* vary according to the slope of the terrain. When the slope is around 25%, the walls of stone are from 5 to 7 metres high, 5 metres if it is 30%, from 2 to 3 metres if 40% and 1.5 or 2 metres if 50% or more. This action has both allowed the retention of the soil, which would otherwise be lost because of the steep slopes, and also removed the stones that were occupying space for cultivation (and used them).

In these terraces, several drains were created in order to minimise the erosion and to canalise the water from rainfall. However, erosion was so intense that stonewalls required constant maintenance work. This huge work led to the total transformation of the environment (Plujà, 2000)

The cultivation of olive trees and especially vineyards started in the $10^{th}$ century and the area cultivated progressively increased until the $19^{th}$ century, when the disease *Phyloxera* arrived in 1879, killing all the vineyards of the Empordà region. This destruction had serious repercussions on the population and led to a radical change of the agricultural landscape.

A general abandonment of the terrain previously occupied by vineyards took place, leading to the cultivation of agricultural fields in the plain, where soils are more fertile and the agricultural machinery has a better access.

Although the agricultural land decreased enormously, some patches of olive plantations remained. Olive was an important crop until February 1956 when extremely cold temperatures killed the majority of the remaining trees, leading to the general abandonment of the cultivated fields in the mountain parts. Nowadays the cultivated surface is less than 5% of the total terrain in the littoral and less than 10% in the interior villages (e.g. Vilajuïga). The economical alternative for the habitants of these rural areas is tourism.

#### 1.4.7.2.3. Farmland activities and periodic wildfires

Grazing activities and fires can be either anthropic or of natural origin. These actions have become one of the most important factors of the modification of the landscape of the peninsula of Cap de Creus.

In this area, grazing practices have taken place since ancient times although the progressive abandonment of the rural houses dedicated to the breeding of herds, led to a significant decrease of these activities. Nowadays, only herds from the Pyrenees are brought from the Ripollès region at the end of November for their stay at this site until the end of the spring season.

Traditionally, shepherds have burnt shrub vegetation in order to obtain pasture for their livestock. This practice affects the vegetation dynamics due to the impediment of the succession of the system. The entry of the herds to burned areas in order to have access to the tender buds has serious effects on the degradation of the vegetation as well as the soil, mainly due to the trampling of the animals. To a large extent, grazing activities in the area are linked to human-induced wildfires. Thus, during the past 25 years, almost the 50% of the fires have been induced.

According to the annual fire-frequency maps, the Cap de Creus has been the most frequently burned area in Catalonia over the past twenty years. The high presence of this agent has been favoured by several factors such as the climatic conditions (frequency of the Tramuntana wind and the severe summer drought), the availability of the fuel (the history of progressive rural abandonment over the 19th century), the inflammable characteristics of the vegetation (constituted by shrubs with a very low water content during summer and rich in volatile substances), its topography (which makes the extinguishing of fires difficult) and probably the increased frequency of ignition due to grazing, visitors, etc (Pons, 2000).

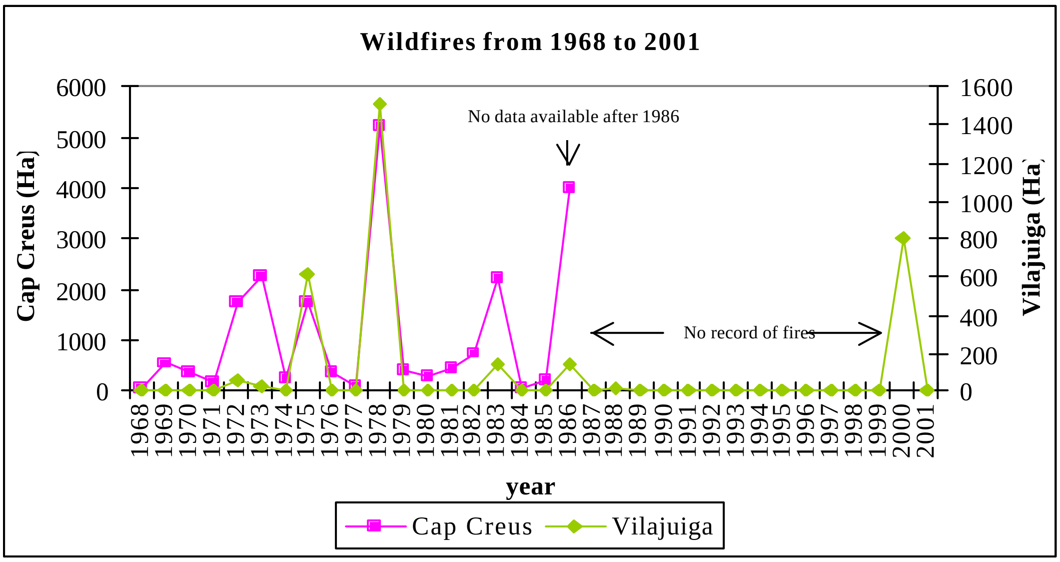


Figure 1.3. Wildfires in Vilajuiga (1968-2001) and in the Cap de Creus (1968-1986)

According to the information from the Department of Environment of the Generalitat of Catalonia government, during the period 1968-1986, a total of 21,041 ha were registered as being burned in the area of the Cap de Creus Natural Park (Figure 1.3).

Some places appear in the register of fires more than 12 times during these years (Fortià, 1993). In this same context, for the period 1968-2001, a total of 3291.1 ha were burned in the municipality of Vilajuiga, showing a frequent recurrence of the wildfires in this area (Figure 1.3).

From this information it can be concluded that the vegetation species currently dominant in the landscape of the Cap de Creus are adapted to fire conditions, having the resources to assure their conservation. As a result of the repeated wildfires, the vegetation succession goes to more regressive stages with the presence of more scrub vegetation and pasture areas.

Some factors directly related to fire have an important role on the evolution of the vegetation. First, the condition of the plants at the time of the fire is important, because its effects in the Cap de Creus show the existence of an autosuccession phenomenon in which the burned communities reach a floristic composition and structure identical to the initial ones after some years. Only a frequent recurrence of the fires could produce the loss of this recuperation capacity and consequently the disappearance of a particular site. Secondly, climatic factors are also of maximum importance. Precipitation events after a fire are essential with reference to the presence and activity of the edaphic micro-organisms which can assure the mineralization of the ashes (Ahlgrin, 1974). On the other hand, torrential rainfalls after the fire, when the soil is unprotected, could provoke considerable erosion problems. In addition to the loss of substratum of the soil, the erosion could imply the removal of seeds and consequently the diminution of the germination of new plants in the burned area. Erosion after a fire can be a factor on the degradation and loss of the vegetation cover (Franquesa, 1995).

To summarize, the interior zone of the Cap de Creus and especially the Serra de Rodes catchment, is a land transformed by human activities leading to the substitution of the primitive vegetation adapted to the edaphic and climatic conditions and the establishment of stationary vegetation communities adapted to the colonization of ancient cultivated terraces and repeated wildfires. The landscape is defined by the monotonous extension of low formations, which in reality is a mosaic of dense brushes, sparse scrubs and pastures of different types. According to the land-use map of Catalonia, published by the Cartographic Institute of Catalonia, this territory is catalogued as matorral.

## 1.5. THESIS OUTLINE

The present chapter aims to set up the desertification issue as the main research framework, to detail the main aim and specific objectives of the thesis, as well as to describe the study area.

Chapter two reviews the processes and causes of land degradation and desertification in order to identify which are the most important in the target area.

Chapter three reviews the utility of indicators on the assessment of land degradation and desertification processes. The most relevant indicators related to land degradation processes are discussed. Based on the previous literature review a set of potential indicators of desertification will be developed and applied in the DIS model.

Chapter four describes in detail the required methodology for achieving the specific objectives and therefore the main aim of the thesis.

Chapter five assesses the results of the research conducted in order to accomplish the several objectives set up in chapter one.

Chapter six focuses on the description of the development of the (DIS).

Chapter seven addresses the sensitivity analysis of the DIS in order to demonstrate the sensitivity of the model simulations to uncertainty in values of model input data.

Chapter eight is based on the validation of the DIS with plot scale data and also on the development of land use scenarios of particular policies addressed to land degradation issues.

Chapter nine summarises the results of the research and draws attention to the major conclusions and future research work.

# 2. LITERATURE REVIEW: PROCESSES AND CAUSES OF LAND DEGRADATION



## 2.1. PROCESSES OF DESERTIFICATION/ LAND DEGRADATION

Desertification is the result of a series of natural processes and of processes due to human and animal pressures leading to gradual environmental degradation (Rapp, 1974). According to the FAO (1984) the main processes of land degradation are: the degradation of the vegetative cover, soil degradation and soil erosion (Figure 2.1).

### 2.1.1. DEGRADATION OF THE VEGETATIVE COVER

Degradation of the vegetative cover refers to the removal or destruction of the vegetation (Poesen, 1995). This destruction is considered the dominant biotic component of the desertification process. Several components of each plant such as the canopy and litter are important in different ways, in terms of their contribution to the overall reduction of soil loss with increasing vegetation cover. Some of the many effects of the vegetation on soil erosion by water are: hydrological effects (conditioning the erosivity of rainfall and runoff), mechanical effects and effects on soil properties.

Therefore, vegetation cover is important as a protection of the soil from erosion by wind and water on the one hand, and as a provider of its organic material to maintain levels of nutrients essential for healthy plant growth on the other. Furthermore, plant roots help to maintain soil structure and facilitate water infiltration. The vegetation canopy controls water input into the soil by affecting interception, stemflow and infiltration rate. It should be stressed that vegetation also modifies a number of soil properties controlling soil erodibility, such as organic matter content or soil structure. Key processes of desertification related to the existing natural or agricultural vegetation can be considered in relation to fire risk and the subsequent ability of the vegetation to recover, erosion protection offered to the soil and drought resistance (Kirkby and Kosmas, 1999).

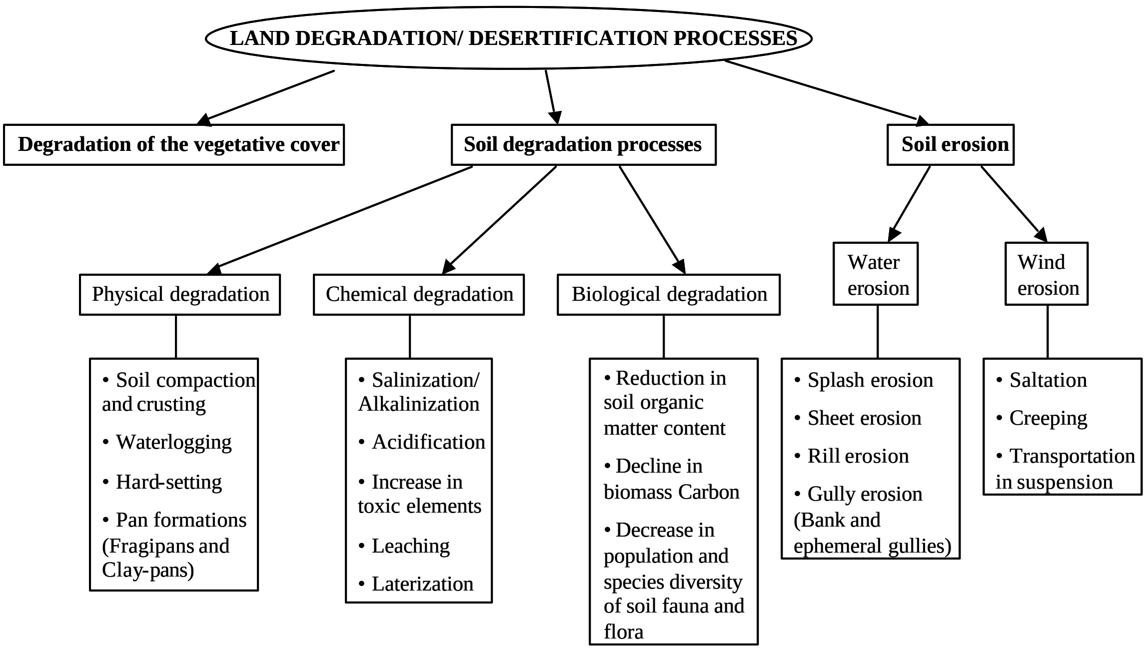


Figure 2.1. Main desertification/ land-degradation processes (after Lal et al., 1989).

### 2.1.2. SOIL-DEGRADATION PROCESSES

Soil is the most basic of all resources and the primary medium for food production. It is also a non-renewable resource over the human time scale. Soil degradation is defined as a diminution of soil quality (and thereby its current and potential productivity), and/or a reduction in its ability to be a multi-purpose resource due both to natural and human-induced causes. Decrease in soil quality implies there have been changes in soil properties and processes that have an adverse effect on the ability of the soil to support life. Soil-degradation processes include chemical, physical and biological actions and interactions that affect the capacity of the soil for self-regulation and its productivity.

### 2.1.2.1. Physical processes of soil degradation

Physical processes of soil degradation lead to changes in soil physical, mechanical, hydrological and rheological properties, which have a negative effect on crop and animal production, farm income and environmental quality (Lal et al., 1989). The adverse changes in soil physical properties include changes in porosity, bulk density, structural stability and permeability of the soil, and it comprises processes leading to a reduction of the macro porosity of the top soil such as surface sealing, crusting and compaction (Poesen and Nearing, 1993). Change in soil structure is a principal effect of physical degradation. Soil structure refers to the macroporosity, stability and continuity of macropores, and to the functional attributes of soil pores to transmit and retain fluids and facilitate root growth. Deterioration of these structural attributes is manifested in different types of commonly observed processes such as sealing and surface crusting, hard-setting, reduction in soil elasticity, decrease in bearing capacity and trafficability, soil compaction leading to impeded root growth, and water imbalance causing poor drainage, frequent drought, excessive overland flow and accelerated erosion. These types of physical degradation are interrelated and lead from one problem to another. The mains forms of this kind of degradation are: soil compaction and accelerated soil erosion, processes that lead to increase water runoff, decrease amounts of available soil water, impairment of seedling emergence and root penetration (Lal et al., 1989).

#### 2.1.2.1.1. Soil compaction and crusting: Natural and/or human-induced physical process of soil degradation

Both compaction and crusting are natural features of arid and semi-arid soils. Compaction and crusting, whether natural or human induced, tend to be greater in soils with a low organic matter and expanding type clay (Dregne, 1991). Crusting, especially, is harmful because the crust reduces infiltration rates and increases runoff and the potentially for soil erosion.

Root-zone compaction is a principal form of physical degradation observed on intensively managed croplands and pastures. Structurally inert soils, containing low organic matter and predominantly low activity clays, are most prone to compaction. The structural stability of these soils is extremely vulnerable to the mechanical forces involved in normal farm operations. Compaction modifies the pore volume and pore size distribution. With the exception of laterization and other forms of chemically related compaction, it is rare that a soil is irretrievably degraded due to compaction. Notable among these are an increase in the volume and rate of overland flow, accelerated erosion, increased wetness and poor aeration, and a decrease in biomass production.

#### 2.1.2.1.2. **Waterlogging: Natural and/or human-induced physical process of soil degradation**

Waterlogging is the process that takes place when the porosity of the soil is entirely filled with water. It is also called the raising of the water table (Mainguet, 1994). Waterlogging affects one or several horizons of the soil, seasonally or permanently. When waterlogging becomes excessive, it may result in degradation, mainly when the water brings salts towards the surface. This soil degradation results from excessive waterlogging, i.e. when excess water in a soil horizon limits the aerobic life. In the evolution towards an anaerobic medium, the microorganisms responsible for biodegradation of organic material are inhibited or destroyed. An accumulation of organic matter occurs resulting from slowing down of mineralization and humification of plant debris. Some of these chemical or structural changes decrease stability and fertility and increase the vulnerability of the medium until irreversible degradation occurs (WMO, 1983).

#### 2.1.2.1.3. Hard-setting: Natural physical process of soil degradation

Hard-setting soils are identified as weathered soils, low in phosphorous, with dense and sometimes weakly cemented surface horizons. Their low fertility has resulted in limited biological activity and soils low in organic matter. Establishment of plants on hard-setting soils is difficult because of the low porosity, particularly with respect to macropores, and the presence of fine laminations in the upper few centimetres (Lal et al., 1989). This low porosity and the limited vegetation result in high runoff of precipitation.

Although hard-setting occurs under native vegetation, it is frequently associated with cultivation. Intensive cultivation or over stocking of pastures may deplete organic matter and lead to the loss of soil structure and dispersion characteristic of hard-setting.

#### 2.1.2.1.4. Pan formations: Natural physical process of soil degradation

A pan is a hard concretionary layer formed at, or beneath, the soil surface (Barrow, 1994). Often pans form just below the cultivation depth; these are sometimes called plough soles. They can restrict roots, making crops or natural cover vulnerable to drought and trees vulnerable to wind-throw. They can also affect drainage leading to waterlogging and salinity/ alkalinity problems. It can be distinguished two different types of pan formations, the fragipan and the clay-pan formation.

*Fragipans* are pedogenic subsurface horizons characterized by medium textures with a high content of silt with very fine sand and low to moderate clay content, high bulk density, very low organic matter content, and brittleness when moist and hardness when dry (Lal et al., 1989). Because of their high bulk density, fragipans have very low hydraulic conductivities. The combination of high bulk density and low hydraulic conductivity causes soils containing fragipans to have limited rooting zones and so, in dry and wet growing seasons, crops suffer more severe climate-induced stresses than soils without such horizons. Erosion of soils containing fragipans is more likely to affect crop yields than similar rates of erosion on soils with deeper rooting zones.

There are millions of hectares of these subsurface pans and they clearly restrict the use of the land where they occur. Since these and similar pedogenic pans restrict root growth, hydraulic conductivity and affect other soil characteristics, pan formation is a form of natural soil degradation which restricts the potential of the soils for plant growth (Lal et al., 1989).

The *Clay-pan* or argillic horizon process occurs when clay, particularly fine clay moves from the surface (A and E horizons), and accumulates in the subsoil (B horizon) or there is in situ a formation of clay in the B horizon, and it results in a physical barrier to roots and to the movement of water (Soil Survey Staff, 1975). This barrier is particularly restrictive if the subsoil is lacking in well-developed soil structural units. In general the degree of degradation attributable to a clay pan soil is a function of the B/A clay ratio and the distance over which the increase in clay takes place (Lal et al., 1989). An increase in clay in the soils would limit rooting, particularly where the soil lacks strong structure.

#### 2.1.2.2. Chemical degradation processes

Chemical degradation processes comprise changes in soil's chemical properties that regulate nutrient activity and capacity, or which maintain a favourable balance among principal nutrient elements, and the accumulation of substances possibly to toxic concentrations. This kind of degradation leads to a reduction in a soil's ability to inactivate toxic compounds, and it includes salinity, alkalinity and acidity. Depletion of major plant nutrients, accumulation of salts and heavy metals in concentrations toxic to plant growth, leaching of bases and accumulation of $Al^{3+}$, $Mn^{3-}$ on the exchange complex, are the principal processes of soil chemical degradation (Lal et al., 1989).

#### 2.1.2.2.1. Salinization and Sodification: Natural or/and human-induced process of soil degradation

Human-induced salinization resulting from the spread of irrigation is a sinister and widespread form of desertification (Goudie, 1990). The transport and distribution of salts within a landscape and in a soil profile reflect the prevailing water balance conditions, and the soil depth to groundwater. Therefore, precipitation and evapotranspiration together with soil-profile characteristics are important for the distribution of salts in the landscape and in the soil profile.

A high concentration of salts in the soil gives rise to saline or alkaline soils. This process consists of the accumulation of soluble salts on or at various depths in the soil profile (e.g. in irrigated areas) and the exposure of saline rocks (Poesen, 1995). The accumulated of salts often includes chlorides, sulphates and carbonates of sodium, magnesium and calcium. Abundant salts, and more than 15% exchangeable sodium, lead to the formation of alkali soils by alkaline hydrolysis (Mainguet, 1994; Lal et al., 1989). Thus, this process is termed alkalinization, broadly synonymous terms are: sodification, sodication, solodization or soda alkalinization, and such soils generally have a pH of 8.5 or more (Barrow, 1994). According to the FAO, a soil is classified as saline when its content in soluble salt exceeds 1 to 2% in the upper 20 cm. In general and depending on plant species, saline conditions are said to have developed when the accumulation of salts has reached a level harmful to plant growth. The increase in soluble salts in the soil increases the osmotic pressure and simultaneously increases the difficulty for plants of utilising water, a state of physiological drought, although some plants are more salt-tolerant than others. If the salty groundwater reaches the soil surface it evaporates and leaves salt crystals on the surface of the fields and if the quantity of salty water reaching the surface is great enough, a relatively impermeable salt crust will form over the soil, diminishing infiltration and natural leaching (Mainguet, 1994).

Salinity or alkalinity in a soil is not constant, it varies from place to place even over short distances, depending on tillage, local soil quality, local topography and distance from surface to the water table. Soil salinity may also vary in time, during the dry season, if there is one, when there is little rain to leach out salts, causing salinity to rise (Barrow, 1994).

#### 2.1.2.2.2. Acidification: Natural and/ or human-induced processes of soil degradation

There are many soils that are naturally acidic even without interference by people. Soil acidity is determined by soil components and the nature and extent of chemical and biological reactions in the soil systems and is largely controlled by the degree of weathering. Weathering and dissolution of parent materials by hydrolysis of $CO_2^-$, followed by leaching of basic cations ($Na^+$, $Ca^{2+}$ and $Mg^{2+}$) with bicarbonate, is the dominant acidification process in nature. Weathering of acid parent materials or oxidation of acid minerals produces additional soil acidity. Organic matter or humus contains reactive carboxylic and phenolic groups, which act as weak acids and dissociate $H^+$ through deprotonation. It is also a potential source of stronger acid $HNO_3^-$, produced by nitrification of organic matter. Aluminium and iron hydrolyses are often major sources of soil acidity for highly weathered soils (Norton et al., 1999). On the other hand, some soils can become acidified due to cultivation and intense cropping. Such human-induced acidification is characterised by rapid oxidation of soil organic matter content and excessive leaching of bases out of the root zone. Acid chemical fertilisers can also produce a considerable amount of acid by nitrification of ammonium fertiliser materials. In addition, removal of basic cations by plants, redox reactions and acid rain may also cause soil acidification to a certain degree.

Leaching and acidification can occur in very humid conditions, in soils of predominantly low chemical activity clays and containing low levels of soil organic matter. The result is the loss of bases and clay that alter the physical and chemical properties of the soil.

#### 2.1.2.2.3. Increase in toxic elements: human-induced process of soil degradation

Toxic element accumulation is a local problem in the arid regions, and it is associated with industrial activity. Heavy metals, such as lead, cadmium, and nickel, are the principal elemental pollutants of soils, primarily because of their toxicity to humans and animals when plants growing on contaminated soils are consumed. Heavy metals, boron and other elements toxic to plants or animals are deposited by smoke, dust or runoff from industrial and mining operations are carried in irrigated water (El-Hinnawi and Hashmi, 1982). Most of the metals stay in the topsoil because of their limited solubility in water and their strong adsorption on soil particles.

The affected area is usually small, by the impact can be considerable. Heavy metal accumulation is practically irreversible. Excess boron can be washed out of the soil but at a heavy cost in water (Dregne, 1991).

#### 2.1.2.2.4. Leaching: Natural and human-induced processes of soil degradation

Leached soils are distributed over most of the climatic zones of the world, excluding arid and semi-arid climates in which evapotranspiration is higher than rainfall for most of the years. Nevertheless, leaching can exist at a dramatic level in the dry ecozones under irrigation. Percolation of water into the soils is high and the profiles are heavily desaturated and acidic (Mainguet, 1994).

Leaching is the removal from within and outside the soils of colloids and the chemical bases associated (WMO, 1983). Leaching leads to the loss of bases such as $Ca^{2+}$, $Mg^{2+}$, $K^{+}$, and $Na^{+}$ and accumulation of $Al^{3+}$ and $Mn^{2+}$, and can be associated with loss of the clay fraction. It consists in the migration of nonsolid substances from the upper horizons to the lower ones. The transfer can be vertical when the accumulation is in the deep levels of soil and oblique when leaching is produced on a slope or in a profile located downstream.

#### 2.1.2.2.5. Laterization: Natural process of soil degradation

Laterization is a general term to describe the process of iron accumulation in soils. It is a process, which at its extreme, involves intense weathering (resulting in a breakdown of all minerals except quartz) and intense leaching of the soil which removes all the soluble salts, much of the silica and some of the iron and aluminium (Lal et al., 1989). Kaolinite is the dominant clay mineral formed as a result of the process. The iron and aluminum in lateritic material has traditionally been considered to be the result of residual accumulation from the original parent material. As a result of intense weathering such soils are very low in the nutrients needed for plant growth. The high iron and aluminum content results in a complex fertility-management problem, and where the iron has hardened into irostone, roots are restricted to the upper portion of the soil.

### 2.1.2.3. Biological degradation processes

Biological degradation processes refer to the reduction in soil organic matter and living organisms and to the rate of vegetational decomposition in the soil. These processes are mainly due to the increase of the humus mineralization rate in the soil (e.g. due to temperature increase; excessive cultivation or burning of crop residues), as well as the reduction of the vegetative cover, and the selective removal of the colloidal fraction of the top soil by water and/or wind. Its rate can be expressed by the reduction of organic matter content of the topsoil in % per year.

##### 2.1.2.3.1. Loss of organic matter (humus): Natural and/or human-induced degradation process

Organic matter (humus) plays two major agricultural roles in soils. One is to supply nutrients to crops upon mineralization, principally nitrogen, phosphorus and sulphur. The other is to form stable aggregates that reduce erosion, compaction, and crusting while increasing aeration and biological activity (Dregne, 1991). Humus is an essential substance, increasing soil porosity, infiltration rates, aggregate stability, porosity, water holding capacity and nutrient-holding capacity. As organic matter is concentrated near the surface, this valuable material is generally the first to be lost.

In dry climates, the loss of soil organic matter generally leads to a reduction in retention of soil moisture and, with this, a decline in vegetation cover, crops or natural plants, which, in turn, leads to increased erosion (Barrow, 1994).

### 2.1.3. SOIL-EROSION PROCESSES

Soil erosion is one of the main desertification processes that interacts with other desertification processes such as the degradation of vegetation, biological and physical soil degradation as well as soil salinization (chemical degradation) (Poesen and Nearing, 1993). It is considered the principal component of land degradation, and it starts a chain reaction that exacerbates all aspects of environmental degradation due to adverse changes in the atmosphere, biosphere, hydrosphere and lithosphere (Lal, 1999).

Soil erosion is defined as the physical detachment, entrainment (or transport), and deposition (or sedimentation) of a soil particle (Mainguet, 1994). As the soil is disturbed, organic matter is reduced and aggregates become fewer stables, and the soil more susceptible to erosion (Darmody and Norton, 1994). As erosion of a soil occurs, it is often chemically degraded because of the selective removal of organic matter and changes in accompanying soil chemical factors.

Soil erosion by water and by wind are the corresponding processes of soil erosion. On the one hand, soil erosion is considered to be natural or geological, when it takes place without the influence of people by the action of wind, water, temperature changes, and biological activity (FAO, 1967). On the other hand, soil erosion is defined as an accelerated process, when the soil becomes exposed to direct wind and/ or water action after a disturbance of the vegetation cover (by cultivation, grazing or burning), and it can be removed at a faster rate than it can be regenerate, resulting in a net loss of soil (FAO, 1967).

#### 2.1.3.1. Water erosion

The water-erosion process refers to the removal of soil by rainfall, runoff or gravity (mass movements). This process is by far the dominant erosion process in the northern part of the Mediterranean.

Water erosion is caused by various sources of water: rainfall, melted ice, irrigation water and rivers and other watercourses (Balba, 1995). Rainfall erosion is more widely spread than other causes. The rainfall characteristics of importance in soil detachment are drop size distribution and the angle and velocity of raindrop impact. The latter depends on whether the rain is wind-driven or not. Actually, kinetic energy of the rain is the principal factor responsible for soil detachment, although momentum is also considered important (Lal, 1990). Rainfall causes soil erosion when rain water flows on the soil surface. Erosion also takes place when water infiltration through the soil decreases. There are four forms of water erosion: splash, interrill, rill, gully erosion and piping (Soil Survey Staff, 1951).

##### 2.1.3.1.1. Splash erosion: Natural or/and human-induced process of soil degradation

Splash erosion takes place when the drops of rain water fall on the soil surface and each drop collides with the soil. Thus, the soil clods are detached into small single particles, and as the raindrop collides with the soil surface, it is fragmented and splashed, carrying the single soil particles away from the spot of collision. Downslope movement is greater than upslope movement (Balba, 1995).

##### 2.1.3.1.2. Interrill erosion: Natural or/and human-induced process of soil degradation

This form of water erosion takes place when (1) the disintegration of the soil clods and their movement are regular, (2) the soil is directly affected by heavy rain, (3) the soil surface is smooth and regularly sloping and (4) the rate of rainfall is more than the rate of water infiltration through the soil, thus water accumulates on the soil surface and starts to flow to the lower areas (Balba, 1995).

##### 2.1.3.1.3. Rill and Interrill erosion: Natural or/and human-induced process of soil degradation

The main difference between both processes is that interrill detachment and transport is aided by raindrop impact whereas rills are turbulent and generally not affected by raindrops (Norton et al., 1999). While interrill processes are important for runoff production, breakdown of aggregates, and clay dispersion, rill processes generally lead to the actual removal of soil from a hillslope or field (Norton et al., 1999).

Raindrop impact is the dominant process in detachment in interrill areas. The interaction between the raindrops and the soil typically leads to a reduction in surface roughness elements and depressional storage. The development of the surface seal with low roughness and depressional storage promotes runoff and transport by rain-impacted flows (Eltz and Norton, 1997). However, the increase in strength of the seal results in a reduction in the material detached and partially offsets the increased runoff and transport capacity (Norton, 1987).

A rill is a shallow linear depression or channel in soil that carries water after recent rainfall (Stocking and Murnaghan, 2001). They are caused by the action of water and they normally occur on a sloping surface where runoff is prevalent because of land use and lack of vegetation. The water starts to flow upon the increase of its amount until it spills over the sides of holes and cracks causing erosion of the sides and bottoms of these channels. These small channels are combined, forming larger but flat channels usually termed rills (SCS, 1963; FAO, 1965).

#### 2.1.3.1.4. Gully erosion: Natural or/and human-induced process of soil degradation

The water runs on the soil surface in small channels towards the major slopes. Its movement and ability to erode the soil is increased, forming gullies (SCS, 1963). The gullies magnify the destructive power of the water because it is concentrated in deep channels and moves faster and at a higher energy than when it moves in thin layers along the slope. Because the water in the gullies contacts the least area of land for a short period of time, the infiltration rate of water in the gullies is low which increases the water flow and its ability to erode the soil. Although it depends on the geographical location, interrill and rill forms of erosion are normally more frequent and more serious than gully erosion. It can be distinguished two different types of gully erosion according to the place where overland flow concentrates: bank and ephemeral gullies (Poesen, 1995).

*Bank gullies* form where concentrated overland flow crosses and an earth bank (i.e. a terrace, a road bank or an exploitation talus) (Poesen 1995). Bank gullies are affected more by processes such as piping and mass movement than by overland flow erosion.

*Ephemeral gullies* form where overland flow concentrate in the landscape (i.e. in natural drainage lines), or along (or in) linear landscape elements (i.e. drill lines, plough furrows, parcel borders, access roads) (Poesen and Govers, 1990). Ephemeral gullies can be held responsible for more than 50% of the total sediment output of small agricultural catchments (Vandaele, 1993).

**2.1.3.2. Wind erosion**

In arid and semiarid regions, the action of the wind upon unconsolidated superficial material also can give rise to several problems where the wind has a high velocity, the soil clods are easily broken down and entrained, and the vegetation is meager and provides only a cover for a slight proportion of the soil surface (Balba, 1995).

Erosion of soil by wind is defined as the removal or deposition of soil materials by wind. The coarser particles of soil carried by the eroding wind usually move close to the ground surface until their movement is arrested by physical obstructions, thus large heaps form.

According to Ambrust (1987), wind erosion can occur whenever (1) the soil is loose, dry, and finely divided, (2) the soil surface is smooth and vegetative cover is sparse or non existent, (3) the field is sufficiently large, and (4) the wind velocity is high enough to move soil. The lighter the wind load of soil particles the greater is its ability to breakdown and move more soil particles. The movement of soil particles occurs according to one or more of the following processes (FAO, 1967; Soil Survey Staff, 1951):

#### 2.1.3.2.1. Saltation: Natural process of soil degradation

The forward movement of wind occurs in a parabola and usually in strong successive waves. The soil particles jump during their movement as they go up and drop several times. The collision of the dropping particles with the soil surface increases their effect on soil aggregates. The circulating movement of wind also increases its ability to detach the soil particles and entrain them. The saltation of soil particles is considered one of the most effective mechanisms of wind erosion. Soil particles having diameters within the range of 0.05 to 0.5 mm move in saltation (Balba, 1995).

#### 2.1.3.2.2. Creeping: Natural process of soil degradation

Winds of slow speed may be unable to uplift soil particles of large sizes, thus these particles may creep along the soil surface. They also may collide with other particles, thus moving them. Winds are not able to carry soil particles if the velocity of their layer contacting the soil surface is less than 0.5 meters per second (Fryberger, 1979).

#### 2.1.3.2.3. Transportation in suspension: Natural process of soil degradation

Small particles of fine sands or smaller move by uplifting in the air in the form of an air suspension. They remain suspended in the air and are not deposited on the land until the wind settles or rain falls (Balba, 1995).

## 2.2. CAUSES OF DESERTIFICATION / LAND DEGRADATION PROCESSES

Desertification in an area will proceed if certain land components are brought beyond specific thresholds, beyond which further change produces effectively irreversible change (Kirkby and Kosmas, 1999). It can be the consequence of a complex mix of hidden and apparent causes (Warren and Maizels, 1977). Therefore, even though there may appear to be similar processes and causes at work in different regions, the reality may be very different, and interpretation needs caution (Spooner and Mann, 1982; Dregne, 1983). Another trap for the unwary trying to unravel the causes of desertification is that there are often feedback mechanisms, such that a physical cause might lead to a human feedback and that to further physical causes or vice versa (Barrow, 1994). The identification of the causes of land degradation must recognise the interactions between different elements in the landscape, which affect degradation and also the site-specificity of degradation. The effect of a land-degrading process differs depending on the inherent characteristics of the land, specifically soil type, slope, vegetation and climate (Stocking and Murnaghan, 2001; Poesen, 1995). An overview of the main direct and indirect causes, natural and/or human-induced of land degradation/desertification processes is presented in Figuere 2.2.

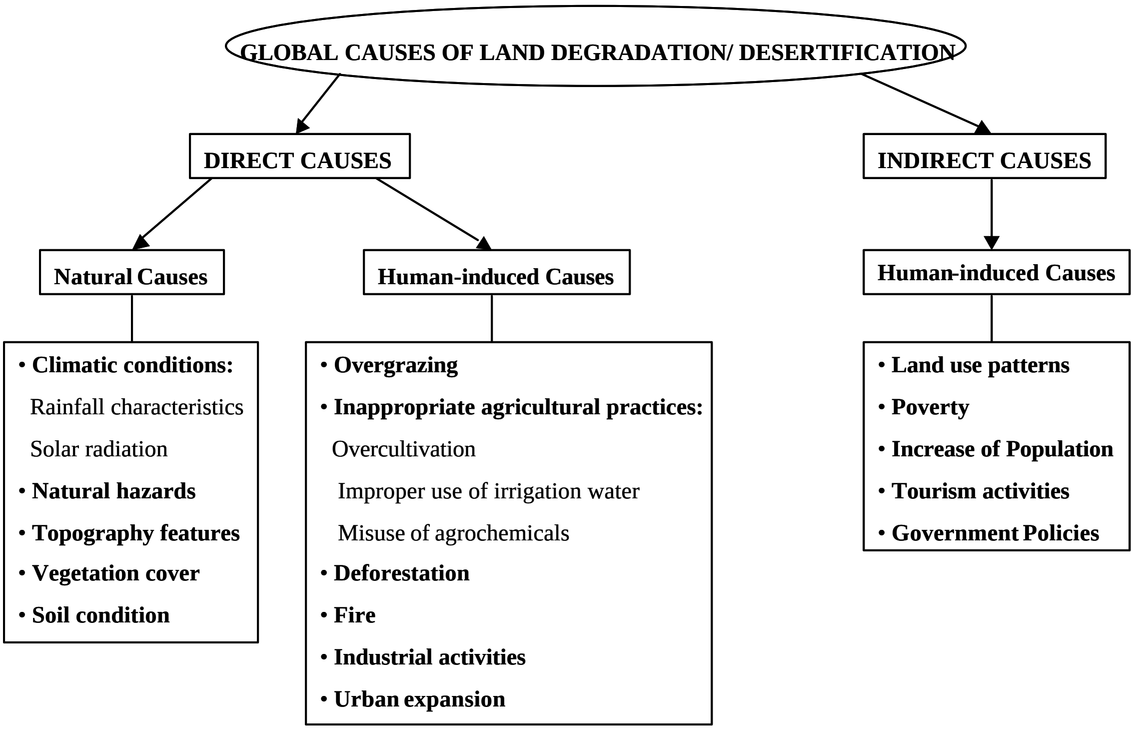


Figure 2.2. Global causes of land degradation/desertification processes (after Lal et al., 1989).

### 2.2.1. Direct causes of desertification/ land degradation processes

Sometimes causes of land degradation are local and relatively simple, sometimes land degradation results from, possibly complex, global changes some of which are at least partly caused by human activity (Barrow, 1994). Factors of land degradation are natural and human-induced agents and catalysts that set in motion those processes that lead to changes in a soil properties and its life-support attributes (Lal et al., 1989).

#### 2.2.1.1. Natural causes

Degradation processes occur without interference by people, but these are broadly at a rate that is in balance with the rate of natural rehabilitation. Land degradation in arid and semiarid regions is in many ways a product of human activity, nevertheless it is significantly amplified by physical factors such as: climate, topography, soil erodibility and the local vegetation status (Pérez-Trejo, 1994).

##### 2.2.1.1.1. Climatic conditions

Climate affects desertification in many ways and it is always unpredictable, and its variability can have dramatic and potentially catastrophic effects on societal systems (Pérez-Trejo, 1994). Thus, instead of focus on the deviations from average conditions, there is a need to focus on the frequency of extreme climatic anomalies, because these are responsible for generating the most serious social and environmental impacts, such as accelerated soil erosion and flood events.

###### 2.2.1.1.1.1. Rainfall characteristics

The intensity and regularity of precipitation, is an important cause of land degradation in arid and semiarid regions. The energetic impact of rainfall at the soil surface modifies the physical soil properties and as a consequence, the soil particles are destabilized, detached and subsequently transported downslope by the running water (Pérez-Trejo, 1994). The pressure exerted by the raindrops also subjects the soil to strong compaction and consolidation of its surface, resulting in an impermeable crust that inhibits infiltration, increases runoff and sediment yield. Thus, the fine fraction of soil and its related organic matter fraction are eroded and it leads to a progressive depletion of soil nutrients, reducing the fertility of the soil.

#### 2.2.1.1.1.2. Solar radiation

High temperatures, in arid and semiarid regions encourage soil evaporation and inhibit drainage resulting in the accumulation of salts at the soil surface. Conversely, the potential for rain is linked to the vegetation cover, which creates a buffer for the solar energy hitting of the soil. If the soil has a high vegetation cover, then the solar heat that has been absorbed will not be re-radiated immediately, but is held by the soil and released during the following day partly as plant transpiration. This increases the moisture content of the air, and feeds the clouds that generate a potential for rain. If the soil gets denuded by human activity, then the solar heat is re-radiated as it hits the soil, and reduces the moisture contribution of plant to atmosphere, thus breaking the cycle, reducing the amount of rain available for vegetation to grow on, and exacerbating further the denudation of the soils (Pérez-Trejo, 1994).

#### 2.2.1.1.2. Natural Hazards

According to Barrow (1994), natural disasters often cause widespread and severe land degradation, although these phenomenon take place in a geological time-scale, and are infrequent on human time-scale. This category includes storms, cyclones, hurricanes, tsunamis, volcanic eruption, and so on. In the short term, these events causes degradation, but, over longer term, without such damage, will not be able to evolve, i.e. the natural regeneration of some tropical forests would be upset, for there would be no clearings and less change for fresh growth. The onset and/ or severity of a natural disaster may owe something to human activity. If development has exceeded environment limits, a natural disaster might speed up what may otherwise have been virtually inevitable decline (Barrow, 1994).

#### 2.2.1.1.3. Topography

The topography of any region exerts a powerful influence on settlement and land-use practices, as well as being a contributory factor in soil erosion. The slope inclination, influences infiltration rates and accelerates runoff, and on the other hand, slope length increases the rate of sediment transport. The steepest slopes often generate mass movements, such as landslides, mudflows and avalanches. These latter, whether by snow, rocks or soil particles, damage vegetation cover and in turn, promote further erosion (Pérez-Trejo, 1994)

In some regions, human activity has over the centuries, fundamentally transformed the topography, in such ways as forest clearance, terracing strategies for cultivation practices and through the modern mechanized farming and the process of urbanization have caused the most radical changes in the local and regional topography. The ever-increasing land needs of urban-industrial expansion mean that large tracts of land are continuously being bulldozed, as construction and road building projects alter the shape of the land, and causing continuous land degradation.

#### 2.2.1.1.4. Vegetation cover

Degradation processes generally begin with the degeneration of plant communities. The degree of soil degradation conditions the vegetation cover, and is in many ways a reflection of the state of this vegetation. The vegetation patterns which cover the landscape, affect the soil in all its dynamics, including water redistribution over and within the soil, as well as their microbiological activity. They are biotic interactions, which generate and maintain soil structure in the top 20 cm layer of the soil through the process of aggregation. This aggregation structure is a strong determinant of the soil's hydrological and biological characteristics (Thornes, 1995), which affect their erosional response of the soil, in terms of soil loss, runoff generation and nutrient losses after a rainfall event.

#### 2.2.1.1.5. Soil condition

Soils are considered as being subject to a set of purely physical processes as well as the result of an evolutionary process and are therefore best viewed as complex, evolving systems (Pérez-Trejo, 1994). The parent material, climate, vegetation and other human activities are responsible for the physical and chemical characteristics of the soils. The combination of these biotic, abiotic and human technological factors result in different degrees of erodibility, factor directly related to the potential degradation of the soils.

### 2.2.1.2. Human-induced causes

There are situations where desertification results from natural causes, but often these are triggered or aggravated by human. There are situations where, if things remained undisturbed by people, degradation may well have been much less and/ or slower to develop (UN, 1977; Goudie, 1981). There are situations where causes are a complex mix of both natural and human-induced causes (Barrow, 1994), and there are situations where purely human causes operate. Irrespective to this, human factors always increase the magnitude of the different processes of land degradation. The main direct human induced causes of land degradation are: overgrazing, inappropriate agricultural practices, deforestation, fire, industrial activities and urban expansion.

#### 2.2.1.2.1. Overgrazing

Overgrazing is widely regarded as a prime cause of desertification (Goudie, 1990). In many areas the increases in free-ranging livestock populations have exceeded the carrying capacity of the land, leading to vegetation degradation and, in turn, to compaction and erosion of the soil (Pérez-Trejo, 1994). In addition, the decline in vegetation precipitated by overgrazing can result in a loss of those plant species, which help to maintain soil structure. Under extreme conditions, overgrazing can actually affect the health of the plant community, even producing a change in species composition. An obvious consequence of overgrazing is an increase in soil erosion, since the gradual denudation of the landscape exposes the soil to erosion by wind and water.

#### 2.2.1.2.2. Inappropriate agricultural practices

##### 2.2.1.2.2.1. Overcultivation

The practice of intensive cropping generally reduces the structure of the soil, and as the soil structure degrades, it is exposed to erosion, e.g. the disastrous effects of cash cropping, particularly on fragile soils, and if it is exacerbated by irrigation methods, which ultimately hasten the onset of land degradation (Pérez-Trejo, 1994).

According to Warren and Agnew (1988), overcultivation is a much greater menace than overgrazing. When the exposure of the surface by clearing for agriculture encourages loss of fertile topsoil or exposes infertile subsoil then resilience is damaged and land degradation takes place. It may be inaccurate to assume that traditional systems of cultivation were perfectly at equilibrium with the environment, but the scale of damage, was certainly less in the past under lower population densities than it is now.

##### 2.2.1.2.2.2. Improper use of irrigation water

Irrigation system have been a feature of dryland agriculture for centuries and in some cases millennia, in effect offering the potential to overcome some of the problems that rainfall deficiencies in drylands pose for crop production. The major problems associated with irrigation schemes are their wasteful use of water, with application rates exceeding possible plant uptake and leading to problems linked to waterlogging, salinization and alkalinization (Thomas and Middleton, 1994).

**2.2.1.2.2.3. Misuse of agrochemicals**

Amending chemical constituents may change the physical condition and soil erodibility by affecting processes important to erosion, including dispersion and flocculation, cohesion, hard-setting, self-mulching, slaking, swelling, and surface sealing (Norton et al., 1999).

The problem of agrochemicals is that their compounds may degrade before, or after application into different harmful substances, some of which can be long lived in the soil. There has not been enough time for testing all the effects of using agrochemicals: chemicals fertilizers, pesticides, herbicides and fungicides, to become apparent. Their long-term effects on the environment, particularly within the soil are not clear. There may be slow, cumulative changes in soil chemistry, structure and populations of soil microorganisms. Agrochemical may have catastrophic effects if they destroy earthworms, termites or pollinating insects.

As an example, in many countries, nitrate fertilizers have become a vital part of agricultural production, because without them, yields would fall to uneconomical levels. The problem is that, nitrogenous fertilizers, phosphate fertilizers and ammonium rich fertilizers can be converted in the soil by Nitrosomonas, bacteria to nitrate compounds. Then, in this form they can leach out to contaminate water bodies and may also be converted by soil microorganisms into nitrogenous gases, which escape to add to the greenhouse effect (Wolman and Fournier, 1987). In Europe it has been realized that that nitrate pollution of groundwater and surface water is a serious problem (Barrow, 1994).

### 2.2.1.2.3. Deforestation

Deforestation can be regarded as an initial trigger to the onset of desertification. From a global perspective, i.e. the forest ecosystems of the Mediterranean basin are among the most degraded, as a consequence of human activity (Pérez-Trejo, 1994). In arid and semi-arid areas, where vegetation is sparse, both trees and woodland play a critical role in maintaining soil stability. The progressive deforestation of land for pasture and crop cultivation, as well as wood for fuel and construction has lead to a relevant reduction of forest covers of its original land area. More recently, the ever-growing needs of industry, and the consumptive demands of a rapidly accelerating tourist sector, have resulted in large scale forest clearance. The ecological consequences of deforestation are considerable since it results in a shift of balance in the water cycle as well as promoting soil erosion, and these disturbances are manifest in increased flooding, landslides and the silting up of rivers and dams (Conacher and Sala, 1998).

### 2.2.1.2.4. Fire

Fire is one of the main factors determining vegetation cover in regions with dry seasons. Soils and faunas are also directly and indirectly affected by fire. Fire can be started by natural causes such as lightning. However, there is some doubt that natural wildfires have sufficient effect on vegetation to actually cause desertification, unless the vegetation has been altered by human (Barrow, 1994). Vegetation is likely to be adapted to natural fires so that it has the ability to withstand burning and/ or re-seed or re-sprout (Naveh, 1990). Therefore, vegetation can be fire adapted and depends on bushfires to maintain itself. Prevention of natural fires may lead to alteration of natural vegetation cover and possibly land degradation. In some regions, human initiates so many wildfires/ bushfires that the recurrence interval between burns becomes so short that vegetation and soil deteriorate, because of their inability to recover their initial conditions, after and between each physical disturbance.

For a while after a bushfire there is likely to be increased runoff and erosion. This decreases as grass and herbs develop, and may even fall below what there was when there was a cover of brush or forest with little understorey, because transpiration losses have grown with the revegetation. The damage following a fire depends on the season in which it struck, the severity of the burn, and the time since the last one. If fire strikes too soon in the growing season, plants may find it difficult to regenerate, as they have not set seed nor have gained strength in their below ground structures. Fire after the growing season will have less long term effect because the vegetation has had a chance to scatter seed and form tubers. Fire followed by heavy rains will tend to lead to erosion; light rains soon after a fire may encourage revegetation enough to reduce wind and rain erosion when heavier storms occur. However, some studies have revealed that soils after low intensity fires may show an increase of infiltration and a decrease of erosion (Cerdà et al., 1995). The severity of burn depends on how much above ground dry matter has accumulated; the more of this there is the hotter the fire will tend to be and the more damage it will do to flora, fauna and soil (DeBooysen and Tainton, 1984).

#### 2.2.1.2.5. Industrial activities

Pathogenic organisms, heavy metals, and persistent organic chemicals are the primary troublesome pollutants of industrial wastes (Dregne, 1991). Some of the wastes go directly, legally or illegally, into rivers, lakes, and oceans. Others are sent to landfill sites, where the soil and groundwater may or may not be monitored. Harmful industrial wastes going through a sewage treatment plant usually end up in the plant effluent or in the residual sludge. Although it highly depends on the type, normally if the wastes are in the effluent, it appears unlikely that they will cause problems whether the effluent is used to irrigate food or nonfood crops (Goldstein, 1977). Where industry produces particularly toxic airborne pollutants such as copper, mercury or nickel smelting, extensive areas may suffer vegetation and soil damage and it is very difficult to rehabilitate. Heavy metals, radioactivity in some toxic compounds may be very slow to break down. The soil therefore has either to be removed, and carefully buried below an uncontaminated layer, with measures to prevent accidental exposure of leaching out of the hazardous compound, or treated chemically to neutralize or leach out the pollution for safe disposal (Hutnik and Davis, 1973).

Another type of pollutant mainly derived from industrial activities is the sulphur dioxide ($SO_2$) emissions, which are a major cause of *acid deposition.* The effects of $SO_2$ are almost certainly enhanced by other human-made pollutants, notably, nitric oxide and nitrous oxide; metal particles from industrial activity; ozone produced by motor vehicle pollution undergoing photochemical reactions while in the air; ammonia given off by sewage, livestock and some type of artificial fertilizer (Pearce, 1986). Acid deposition can degrade land in a number of ways (Barrow, 1994): (a) damage to plants and/or animals, (b) direct alteration of soil chemistry/structure, (c) alteration of plant metabolism, (d) alteration of metabolism or species diversity of soil microorganisms leading to change in fertility/soil chemistry.

#### 2.2.1.2.6. Urban expansion

The impact of urbanization is not only felt through the physical presence of buildings. Demand for fuel wood may lead to damage to forests for hundreds of kilometres, and the lure of the city may draw farm labour from the land for thousand of $km^2$ and thereby upset efficient land use causing degradation (Barrow, 1994).

### 2.2.2. INDIRECT CAUSES OF LAND DEGRADATION/ DESERTIFICATION

#### 2.2.2.1. Land-use patterns

The current land use of any region is mainly the result of long-term historical, social, economical, cultural, political, climatic, environmental and even religious factors, all of which are implicated in the changes of ownership and tenure, population growth and urban-industrial development. One of the most significant land use changes during the last century in all the Mediterranean belt, has been the abandonment of the hilly agricultural areas, mainly due to their low fertility, shallow soils and steeper slopes, leading to the desertization of these rural areas. On the other hand, there has taken place the intensification of the agriculture (cash crops) in the plains, and the use and/ or misuse of heavy machinery, fertilizers and agrochemicals (Pardini et al., 2000).

#### 2.2.2.1.1. Land abandonment

Land abandonment may lead to a deterioration or improvement of the soils, depending on the particular land and climatic conditions of the area. Hilly areas that can support sufficient plant cover may improve with time by accumulating organic materials, increasing floral and faunal activity, improving soil structure, increasing in infiltration capacity and therefore, causing a decrease in the erosion potential (Kosmas et al., 1995). In cases of poor plant cover, the erosional processes may be very active and the regeneration of these lands may be irreversible from a human temporal scale. In cases of land partially covered by annual or perennial vegetation, the remaining bare land with soils of low permeability creates favorable conditions for overland flow, soil erosion and land degradation (Kirkby and Kosmas, 1999). The abandonment of terraced land in the Mediterranean region has taken place during the second part of the 20$^{th}$ century, mainly because of the difficult accessibility of the agricultural machinery, and low incomes. Thus, these agricultural structures, have progressively collapsed, causing a rapid removal of the soil by runoff water, apart from where the stone walls are protected by the roots of fast growing shrubs and trees (Kirkby and Kosmas, 1999).

#### 2.2.2.1.2. Tourism activities

Tourism cannot be construed as a direct cause of desertification; nevertheless, it exerts a significant impact on the environment, particularly with respect to land use patterns and water resources availability (Pérez-Trejo, 1994). The more immediate changes in land use are shifts in crop choices to cater for tourist tastes, replacing more traditional crops, but less evident and probably more impacting, is the change in water allocation, which can impair access o water for agriculture or other economic activities, or drive water prices up, forcing costs of production up to levels that only capital-intensive activities such as tourism can absorb.

The direct impact of tourism on landscape resources, results from the continuous need for increased accommodation from an ever-growing tourist sector. Thus, hotel structures are built to gratify a perceived set of tourist wishes, and these construction projects and the land area they occupy are realized at the expense of potential agricultural areas. Perhaps an even more serious consequence of the ever-increasing development of tourism is the serious reduction in water availability due to the excessive water consumption by hotel and leisure complexes. Thus, the resulting lowering of the ground level through over-pumping leads to a "social waste-land" and ultimately to the abandonment of cultivated land (Pérez-Trejo, 1994).

#### 2.2.2.1.3. Poverty

Poverty affects how land users manage their land. It reduces the options available, ruling out some conservative practices because they require too much investment of land, labour and capital (Stocking and Murnaghan, 2001). Poor people generally have no choice but to opt for immediate benefit, very often at the expense of long-term sustainability (Barrow, 1994). Therefore, if a community is too poor to raise the capital needed for restoration of degraded land, then degradation is likely to continue and accelerate (Warren and Agnew, 1988). Thus, poverty induces land degradation, which, in turn, reinforces poverty leading to further land degradation and so on (UN Centre on Transnational Corporations, 1985). Nowadays, this problem has been recognized by the World Bank as a necessary policy in the Third World (Hopper, 1988):

*"Poor people cannot easily postpone immediate consumption for future returns. Nor will they ignore the pressing needs of the moment if these can be met from their limited resources, even if the use of these resources jeopardizes their longer term viability"*.

Another view of this problem is the fact that poverty does not necessarily cause land degradation. Stocking and Murnaghan (2001), based on their field work experience, affirmed that the reality shows that is not true that poor farmers are land degraders, and rich are conservers, because in some cases, poor farmers have been reported to invest more in their land than the rich, probably because they are almost wholly dependent of their land. Poverty therefore, is an ambiguous factor, which needs careful analysis and interpretation in its effect on land degradation.

#### 2.2.2.1.4. Increase of population

Population increase has been one of the most frequently cited causes of land degradation (Barrow, 1994). However, this neglects the fact that there are densely populated regions which produce relatively little erosion or environmental damage, while some areas with low population have been responsible for a great deal of damage. According to Clark and Munn (1986), if population increase has that effect, the impact double edged: a simultaneous increase in demand made upon the environment in order to support growing numbers of people, and a destruction of the resource base. There has been debate on the relationship between population density, growth rate and agricultural development, particularly the intensity of farming (Carlstein,1982). The relationship is by no means a simple one, put crudely it has been claimed that, if a population does not grow enough, or grows too fast for agriculture to respond, then production is likely to remain extensive, and may cause land degradation. If population grows, but not too fast, then intensive agriculture, possibly causing less land degradation, may result (Barrow, 1994).

What is more important in terms of land degradation, is the rate of change of population and settlements patters, instead of the exponential population increase (Pérez-Trejo, 1994).

#### 2.2.2.1.5. Government policies

There are specific policies at a local, regional and more global scale, related to the land protection such as policies supporting terracing, policies favouring extensive agriculture, coastal protection policies, and so on, but their effectiveness depends on the degree to which they are enforced. From the European Union for example, trough the Communitarian Agricultural Policy (CAP), many subsidies are promoted to farmers from determined regions, for investing on specific priority agricultural cultivation, taken into account the need for being competitive with world markets. But these actions lead to intensification of production, localized and concentrated, which can provoke outward migration of the unemployed to urban areas, extended fields to accommodate larger machinery, leading to soil degradation, making farmers even more dependent on mechanization and agrochemicals to maintain yields, and leading to continuous land degradation in the medium to long term.

The problem to combat desertification efficiently from the Governments is the current lack of synergy between the efforts of the scientific community, the policy and decision-makers and the local affected populations (Kosmas, 1998).

### 2.2.3. MAIN SPECIFIC CAUSES OF LAND DEGRADATION/ DESERTIFICATION PROCESSES

In this section, it has only been taken into account the most relevant causes of the main land degradation/ desertification processes.

#### 2.2.3.1. Degradation of the vegetative cover

The degradation of the vegetative cover of the soil may have both climatic and/or anthropic origin (Poesen, 1995). Some of the human-induced causes leading to this degradation process are: forest removal by logging, bush fires, burning of crop residues, overgrazing and harvesting. The main natural causes affecting this process are on one hand, the climatic conditions, and concretely aridity, which leads to a water stress state causing a substantial reduction on the vegetation cover, and on the other hand the soil conditions, soil properties such as soil depth or organic matter content, which have a direct relationship with the vegetation cover (Kirkby and Kosmas, 1999).

#### 2.2.3.2. Soil-degradation processes

Soil degradation is defined as the loss of actual or potential productivity or utility as a result of natural or anthropic factors (Lal, 1993; 1997). It is considered a global threat (Lal and Stewart, 1990), and it has strong impacts on food and energy resources (Pimentel et al., 1976; Lal, 1988a) and environments (Lal, 1997) and the greenhouse effect (Lal et al., 1995a,b; 1997a,b). According to Lal (1999), soil degradation affects all aspects of human society through its adverse impacts on: (1) agricultural productivity and returns per unit input of the essential resources; (2) environmental quality including loss of biodiversity; (3) income, caloric intake, human nutrition, health, and standard of living; (4) social and political stability and equitable distribution of wealth

##### 2.2.3.2.1. Physical processes of soil degradation

The main factors responsible for physical soil degradation are: deforestation, overcultivation (intensive row-cropping and plowing), excessive wheeled traffic and physical soil manipulation (Lal et al., 1989) (Figure 2.3).

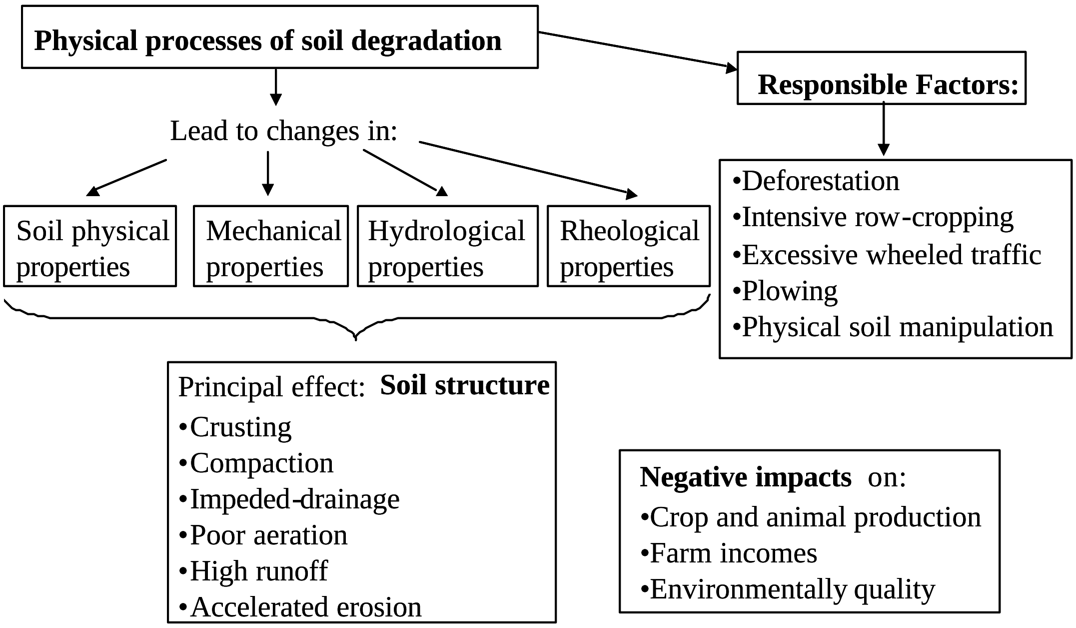


Figure 2.3. Physical soil degradation: main causes and processes (after Lal et al., 1989).

#### 2.2.3.2.1.1. Soil compaction and crusting

Compaction and crusting can be caused by heavy agricultural machinery when it packs the soil and causes plow pans to form, by lying the bare soil and exposing more of the surface to the aggregate-destroying impact of raindrops, and by reducing the organic matter content of the soil (Dregne, 1991). On the other hand, excessive trampling by livestock may also cause soil compaction, which in turns lowers crop yields, decreases the rate of waste infiltration and increases runoff and erosion. Surface crusting, promoted by exposing the soil to the impact of raindrop impact through inappropriate cultivation practices and overgrazing, also increases runoff and erosion and interferes with seedling emergence (Goudie, 1990).

#### 2.2.3.2.1.2. Waterlogging

According to Mainguet (1994), the causes of the waterlogging process are:

- Natural poor external drainage linked with the topography: depression or low-lying areas without natural outlets.
- Climatic factors may also be a determinant in areas of high seasonal rainfall in a short rainy season with excess water inflow in the case of exceptional flooding, and in the case of degradation of the plant cover, which can result in a large increase of the runoff rate
- Poor internal drainage may also induce waterlogging. A clayey texture, an impermeable horizon, compact iron or aluminum hydroxides can be responsible. Salinization can occur after waterlogging.

On the other hand, Barrow (1994), quoted the main causes of waterlogging as follow:

- Altered land use may upset an existing precipitation-evapotranspiration relationship so that water tables rise, even quite subtle changes of plant cover may have this effect;
- Road and rail construction may obstruct drainage;
- Poorly lined irrigation canals and/ or reservoirs may leak and raise local or regional water tables;
- Irrigation schemes may "leak" water raising local water tables, there are large areas of the world where waterlogging/ salinization due to this "leakage" has reduced crop yields to levels below that obtained before irrigation development;
- Rising sea levels or altered base levels, the latter often due to reservoir or barrage construction or inter basin water transfer may cause waterlogging.

#### 2.2.3.2.2. Chemical processes of soil degradation

The main factors responsible for chemical soil degradation are: intensive cropping with no or low nutrient input, disposal of industrial, human and animal wastes, irrigation in arid/semiarid regions using water of poor quality, and accumulation of industrial by-products and gaseous emissions through aerial deposition by “acid rain” and atmospheric circulation (Lal et al., 1989) (Figure 2.4).

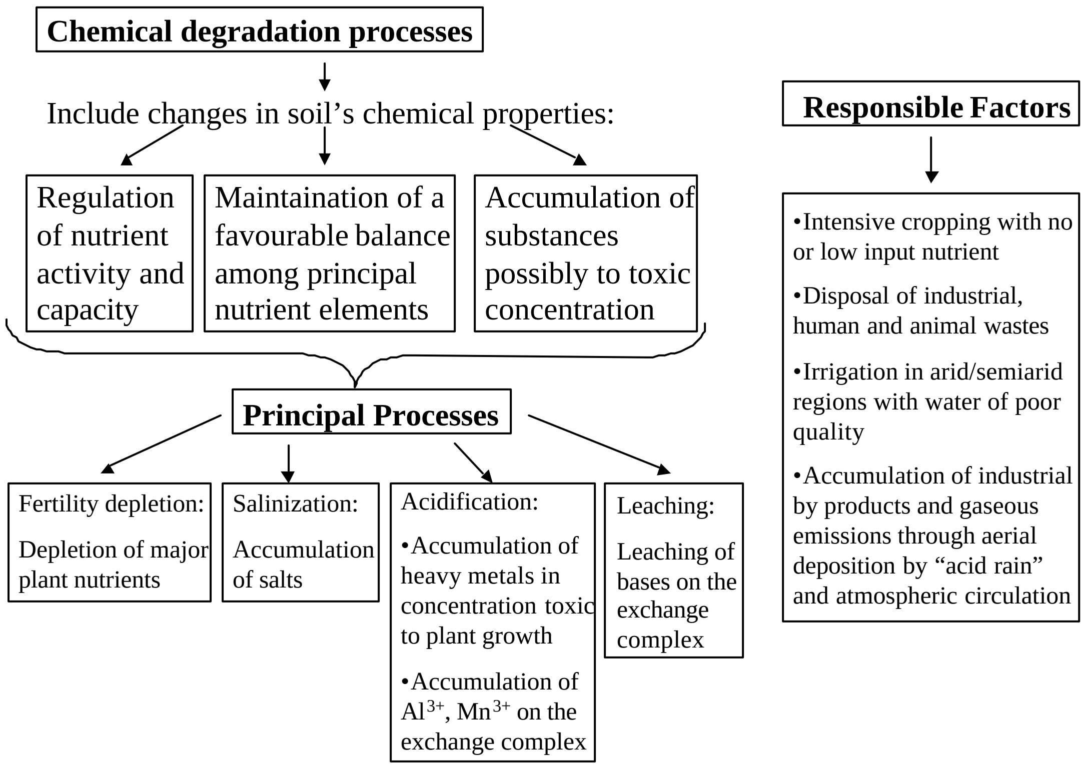


Figure 2.4. Chemical soil degradation: main causes and processes (after Lal et al., 1989)

### 2.2.3.2.2.1. Salinization and sodification

Salinization can result from vegetation clearance, as removal of native forest of bush vegetation allows a greater penetration of rainfall into deeper soil layers which causes groundwater levels to rise, creating seepage of sometimes water in low lying areas (Goudie, 1990). In coastal areas overpumping of aquifers cause seawater incursion. Fresh water is rapidly replaced by salt. A comparable situation can arise in coastal deltas if upstream damming of a river reduces the flow of fresh water to the delta.

In arid and semiarid regions, it is very common that excessive soluble salts accumulate in the root-zone in concentrations toxic to plant growth (Lal et al., 1989). On one hand, it can be derived from a general decrease in precipitation and/or an increase in evapotranspiration, causing an increase in the area of soils affected by saline or sodic conditions. This is because regions with high evaporation rates, capillary rise is accelerated and salts accumulate residually, where drainage is more or less absent. On the other hand, it can also be the result of irrigation practices without adequate drainage and water with a high mineral content: the water is transpired while the minerals increase in concentration. If the solution becomes more concentrated than the solute concentration of the plant cells, water is unable to pass into the plant and a reverse movement of water and nutrients can occur, causing wilting and death to the plant. The increasing concentrations of salts result in radical changes in the water economy of the soil, creating a potentially adverse ecological environment for native vegetation or agricultural crops leading to desertification. According to Barrow (1994), there are four main reasons why irrigation causes salinization/alkalinization: (1) leakage of water from supply canals; (2) over application of water, (3) inadequate provision of drainage and (4) inadequate application of water to leach away salts. Irrigation, or rather poorly designed and/or poorly managed irrigation is considered to be a major cause of salinization/alkalinization in developed and developing countries.

**2.2.3.2.2.2. Acidification**

Acid deposition has become a problem in nations with industrial development and/ or widespread use of coal for fuel. The problem of air pollution related acidification is spreading. Changes in crops and land use may also cause soil acidity changes, as may careless application of fertilizers (Barrow, 1994).

### 2.2.3.2.2.3. Increase in toxic elements

Toxic element accumulation is a local problem in the arid and semi-arid regions (Dregne, 1991). There are many toxic compounds that may build up in soils naturally or as a consequence of development, i.e. boron and selenium are naturally abundant in some soils and ground waters; human activity may bring them to the surface or increase the level of contamination. These elements toxics to plant and animals can be deposited by smoke, dust, or runoff from industrial and mining operation s or are carried out by irrigation water (El-Hinnawi and Hashmi, 1982). Boron, heavy metals, pesticides, herbicides, disease and pest organisms, radioactive compounds and many other contaminants may degrade soil following human activities ranging from warfare to sewage effluent disposal (Barrow, 1994). The affected area is usually small, but the impact can be considerable.

### 2.2.3.2.2.4. Leaching

An excess quantity of water due to precipitation, irrigation or runoff in relation to the speed of infiltration causes stagnation at the surface. In the case of flat topography, a slow infiltration causes leaching in the depth of the soil and reprecipitation when water becomes stagnant (Mainguet, 1994).

### 2.2.3.2.3. Biological processes of soil degradation

The causative factors for reduction in soil organic matter content are: intensive row-cropping, ploughing and other operations leading to mechanical soil disturbance, accelerated soil erosion, excessive applications and indiscriminate use of pesticides, and/or contamination with industrial wastes, soil and crop management that leads to extremes of soil temperature and/or soil-moisture conditions (Lal et al., 1989) (Figure 2.5).

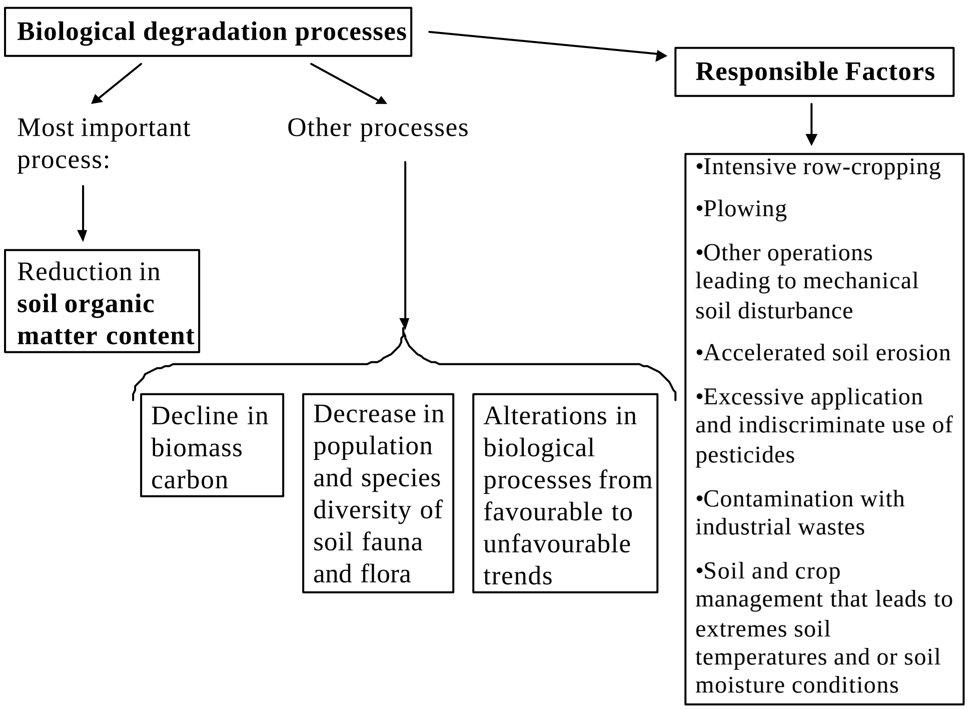


Figure 2.5. Biological soil degradation: main causes and processes (after Lal et al., 1989).

### 2.2.3.2.3.1. Loss of Soil Organic Matter

A pronounced decline in soil organic matter content is usually associated with intensive cropping, summer fallow, or absence of crop rotations with soil-building crops (Dregne, 1991), and it may also result from removal of crops, fodder, wood fuel and dung and bushfires, stubble-burning, alteration of soil drainage, or tillage that accelerates oxidation of organic matter (Barrow, 1994). In rangelands, overgrazing and tree and shrub cutting have a similar effect on organic matter. A soil low in organic matter is almost inevitably lacking in nutrients, and generally leads to the reduction in retention of soil moisture and, with this. Additionally the soil becomes more susceptible to increased erosion, compaction, crusting and runoff (Science Council of Canada, 1986).

A decline in vegetation cover, crops or natural plants, which, in turn, leads to increased soil erosion. A feedback can thus arise, as organic matter/ moisture in soil falls, so plant cover declines and thus renewal of organic matters in soil is further reduced (Barrow, 1994).

### 2.2.3.3.Soil-erosion processes

Soil erosion depends on climatic, topographic, soil-formation, and land use factors (Hudson, 1971). Several factors bring about accelerated soil erosion. Localized high population densities in rural areas have become what is probably the most significant force causing erosion in cultivated lands. High population density, alone does not necessarily lead to land degradation. It is only when people bring about overexploitation of marginal or unsuitable lands that the problems arise. The destruction of the vegetative cover is the number one contributor to soil erosion (Dregne, 1991). Plowing does the best job of laying the land bare. Overgrazing and excessive tree cutting increase the susceptibility of the land to water and wind erosion but have a minor effect compared to cultivation. In fact overgrazing and tree cutting seem to have only a modest effect on erosion unless affected land is on fairly steep slopes.

#### 2.2.3.3.1. Soil erosion by water

According to Lal (1990), the main causes of water erosion are described as:

(a) Rainfall quantity, frequency, and distribution among the year seasons
(b) Gradient and length of soil slope
(c) Kind of vegetative cover, its residues, and proportion of the area covered by plants
(d) The cultivation practices, which decrease the surface flow
(e) Water runoff in arid regions is characterized by flash floods having a high velocity but short duration. This is due to the nature of rain, which falls in short torrents and to the low water infiltration through the soil.
(f) The soil properties, which affect the water infiltration such as particle-size distribution, structural stability, and crusting

On the other hand, Poesen (1995) described the main causes of water erosion as follow: climate, soil condition, topography, and vegetation (natural causes), and human-induced causes.

**(a) Climate:** Physical degradation of the topsoil as well as interrill and rill erosion are controlled, to a large extent, by the kinetic energy of the falling raindrops. Kinetic energy of rainfall increases with rainfall intensity because mean drop size, and hence fall velocity of the raindrops, increases with increasing rainfall intensity.

**(b) Soils:** Erodibility is defined as the property expressing a soil's susceptibility to erosion. Soil erodibility is complementary to the concept of resistance of a soil to erosion.

**(c) Topography:** Topography has a strong control on the rate of soil erosion by overland flow. Overland flow intensity, expressed by for instance flow shear stress, and strongly depends on the slope of the surface as well as on local catchment area or slope length.

**(d) Vegetation:** In general, vegetation cover tends to reduce the erodibility of the topsoil. Soil loss due to water erosion generally decreases exponentially with increasing vegetation cover. The presence of vegetation at the soil surface affects a number of hydrological processes conditioning the erosivity of rainfall and runoff.

**(e) Human-induced causes:** Human can modify the intensity of water erosion by altering one or more of the natural factors: i.e. reduction of vegetation cover, increasing soil Erodibility (i.e. due to a human-induced organic matter decrease or due to removal of rock fragments in the soil top layer). In most cases, this will lead to an accelerated soil erosion rate.

### 2.2.3.3.2. Soil erosion by wind

Wind erosion can result from mismanagement of cropland, overgrazing of pasture land, and excessive cutting of trees and bush cover.

#### 2.2.3.3.2.1. Climatic factors

The relationship between the climatic conditions and wind erosion depends on the intensity and recurrence of wind, rainfall, evaporation, and the soil surface moisture during wind blowing. The moist soil is usually less movable than the dry soil. When the soil moisture reaches a percentage less than the wilting point, wind starts to erode the soil and its ability increases with the increase in its velocity. Wind is a main factor in moving the soil particles by virtue of its energy. Several factors are involved in the relation between the ability to transport soil particles and various atmospheric variables, among which are wind velocity, turbulence, gustiness, shear forces, humidity, and temperature (Middleton, 1990).

#### 2.2.3.3.2.2. Soil factors

The erodibility of soil by wind depends on the detachability of single soil particles or aggregates from the soil and on the size of these particles and aggregates. The smaller the soil particles the easier to move with wind.

According to Chepil and Woodruff (1963), soil erodibility depends largely on its mechanical stability, which they defined as “the resistance of dry soil to breakdown by a mechanical agent such as tillage, force of wind or abrasion from wind blown material”. This mechanical stability depends on size, density, and shape of the soil individual particles.

The sandy soils are made up of single particles, thus the size of the particles is the factor that affects the ease of entrainment. The large sand particles do not move with wind because of their weight. When the particle diameter decreases toward 1mm, its mobility increases. With further decreases to less than 0.1mm, attraction between particles increases until the particles become resistant to movement and the breakdown becomes more difficult, thus the erosion decreases (Balba, 1995).

#### 2.2.3.3.2.3. Roughness factor

Soil roughness is the result of ploughing the fine textured soils or farrowing the coarse textured soils using agricultural machines. It depends on the tillage implement. Soil roughness increases soil resistance to wind erosion by catching the soil particles suspended in the air behind the furrows' ridges. The waves of the soil surface catch the soil particles regardless of the wind direction while the furrows have the same effect angle suits the wind direction (Fryreer, 1987).

Wind erosion occurs mainly right after seeding because the culture practices to prepare the seedbed leave the soil surface smooth and powdered, thus easily eroded. Seeding in the bottom of furrows protect the soil from wind erosion (Balba, 1995).

#### 2.2.3.3.2.4. Vegetation cover

The growing plants as well as plant residues in the field help to protect the soil from wind erosion. They decrease the wind velocity at the soil surface and protect areas with loose particles (Balba, 1995).

#### 2.2.3.3.2.5. Field width

Increasing the number of entrainment and collision of the soil particles with the soil surface, their ability to detach more soil particles increases, thus wind erosion increases.

Accordingly, the total amount of soil entrained by wind depends on its movement distance across the field or on the width of the field (Ambrust, 1987).

## 2.3. SUMMARY

The main processes of land degradation and desertification have been revealed as three: the degradation of the vegetative cover, soil degradation, which may be physical, chemical and/or biological, and soil erosion (by water and/or wind).

In relation to the causes of land degradation, they may be classified into direct (e.g. climatic conditions, vegetation cover and fire), and indirect causes (e.g. land abandonment and government policies). The former may also be divided into natural and/or human-induced causes, whereas the latter is mainly related to land-use patterns. Moreover, the description of specific causes of land degradation processes has been conducted and several factors such as soil erosion, loss of soil organic matter and the degradation of the vegetative cover, are revealed as being both, processes and causes of land degradation.

From this general classification the identification of the main processes and causes of land degradation in the target area may be carried out. To a great extent, the assessment of these processes will be conducted on the basis of indicators of land degradation, believed to be an easy tool for information and management.

# 3. LITERATURE REVIEW: ASSESSMENT OF DESERTIFICATION INDICATORS



## 3.1. INDICATORS OF DESERTIFICATION: AN EASY TOOL FOR INFORMATION AND MANAGEMENT

### 3.1.1. THE UTILITY OF INDICATORS: A GENERAL APPROACH

An indicator may be easy to measure and summarize in shorthand the effects of complex processes that are more difficult to measure or observe (Landres, 1992; Harris et al., 1996). Its purpose is to show how well or bad a system is working. If there is a problem, an indicator is useful in determining what direction to take to address the issue. Indicators can also be useful as proxies or substitutes for measuring conditions that are so complex that there is no direct measurement. According to Winograd (1997), in general, indicators should be useful to: (a) determine the condition of, and change in, the environment in relation to society and the development process; (b) diagnose the actual causes and effects of existing problems that have been detected, in order to elaborate responses and actions, and (c) predict future impacts of human activities on the environment and society to determine future and/or alternative strategies and policies.

On the other hand, according to Tunstall (1992, 1994), the major functions of indicators are: (a) to assess conditions and trends; (b) to compare across places and situations; (c) to assess conditions and trends in relation to goals and targets; (d) to provide early warning information; and (e) to anticipate future conditions and trends.

In summary, desirable indicators are variables that summarise or otherwise simply relevant information, make visible or perceptible phenomena of interest, and quantify, measure, and communicate relevant information. In addition, some indicators may be used to evaluate a condition or phenomenon. Indeed, it is maintained that one of the essential functions of indicators is to quantify. Moreover, indicators can also related to qualitative phenomena.

### 3.1.2. DEFINITION OF AN INDICATOR

A wide survey of publications on environmental indicators shows different meanings assigned to the indicator concept (Gallopin, 1997). An indicator has been defined as a variable (Chevalier et al., 1992; Holling et al., 1978), a parameter (OECD, 1993; Bakkes et al., 1994), a measure (McQueen and Noak, 1988; World Bank, 1995; Dever, 1979); a statistical measure (Tunstall, 1992); a proxy for a measure (McQueen and Noak, 1988); a value (OECD, 1993; Bakkes et al., 1994); a meter or measuring instrument (Adriaanse, 1993); a fraction comparing a quantity (the numerator) with a scientifically or arbitrarily chosen measure (the denominator) (Adriaanse, 1993); an index (Hammond et al., 1995); a subindex or component of an index (Ott, 1978; Adriaanse, 1993; Hammond et al., 1995); a piece of information (Bakkes et al., 1994); a single quantity derived from one variable and used to reflect some attribute (Ott, 1978); and empirical model of reality (Hammond et al., 1995); a sign (Marcus, 1983).

Thus, in principle, an indicator could be either a qualitative (nominal) variable, a rank (ordinal) variable, or a quantitative variable. Therefore, the most important feature of indicators compared to other forms of information is their relevance to policy and decision-making (Gallopin, 1997).

### 3.1.3. INDICATORS FOR LAND DEGRADATION ASSESSMENT

Land degradation is difficult to comprehend in its totality. The productive capacity of the land cannot be assessed simply by any single measure. Therefore, to make assessments of land degradation viable, indicators of its processes and effects have to be used. In the context of desertification, indicators are variables that may show that land degradation has taken place; they are not necessarily the variables controlling the actual degradation itself (Stocking and Murnaghan, 2001). These indicators may be drawn from any aspect of how quality of land degrades. Since there is much interlinkage between the various types and manifestations of land degradation, indicators give us a powerful tool for overall assessments (Stocking and Murnaghan, 2001). According to the United Nations Convention to Combat Desertification (UNCOD, 1978), desertification indicators should provide users with tools, in particular bearing in mind the special requirement of local communities and decision-makers. Thus, the Convention considers indicators as appropriate instrument to provide operational support to a wide range of activities such as estimating, assessing, mapping the extent of desertification, as well as determining the causes, quantifying the impacts, justifying expenditure for mitigation measures and monitoring the efficiency of the measures undertaken (Bouma and Imeson, 2000).

### 3.1.4. CHARACTERISTICS OF AN EFFECTIVE INDICATOR

An indicator is not the same thing as an indication, which is generally not quantifiable, but just a vague clue. Effective indicators, in addition to being quantifiable are characterised by four basic features:

**(a) Relevance:** An indicator must be relevant, that is, it must fit the purpose for measuring. It shows something about the system that is needed to be known, for its assessment.

**(b) Understandability:** An indicator must be understandable, even by people who are not experts. You need to know what it is telling you, in order for you to know when action is needed.

**(c) Reliability:** An indicator must be reliable. You must trust the information that the indicator is providing. An indicator is only useful if you know you can believe what it is showing you. Reliability is not the same as precision. An indicator does not necessarily need to be precise; it just needs to give a reliable picture of the system it is measuring.

**(d) Accessibility of data:** Indicators must provide timely information. They must give you information while there is time to act. In order for an indicator to be useful in preventing or solving a problem, it must give you the information while there is still time to correct the problem. The information is available or can be gathered while there is still time to act.

On the other hand, according to Adriaanse (1993), OECD ((1993), and Tunstall (1994), there are certain universal requirements or desirable properties that indicators should meet, such as:

- ✍✍The values of the indicators must be measurable (or at least observable).
- ✍✍Data must be either already available or they should be obtainable (though special measuring or monitoring activities).
- ✍✍The methodology for data gathering, data processing, and construction of indicators must be clear, transparent and standardized.
- ✍✍Means for building and monitoring the indicators should be available. This includes financial, human and technical capacities.

- The indicators or sets of indicators should be cost effective, an issue which is often overlooked.
- Political acceptability at the appropriate level (local, national, international) must also be fostered. Indicators that are not acceptable to decision-makers are unlikely to influence decisions.
- Participation of, and support by, the public in the use of indicators is highly desirable, as one element of the general requirement of participation of the broader society in the quest for sustainable development.

The CSD (United Nations Commission on Sustainable Development), used the criteria based on the relevance to Agenda 21 and all aspects of sustainable development (Fogh Mortensen, 1997) to determine the most important indicators features for this framework:

- Primarily national in scale or scope (countries may also wish to use indicators at state and provincial levels).
- Relevant to the main objective of assessing progress towards sustainable development.
- Understandable, that is to say, clear, simple and unambiguous.
- Realizable within the capacities of national governments, given logistic, time, technical and other constraints.
- Conceptually well founded.
- Limited in number, remaining open-ended and adaptable to future developments.
- Representative of international consensus, to the greatest extent possible.
- Dependant on data that are readily available or available at a reasonable cost to benefit ratio, adequately documented, of well known quality, and updated at regular intervals.

## 3.1.5. SPATIAL AND TEMPORAL SCALE CONSTRAINTS TO THE CHOICE OF INDICATORS

### 3.1.5.1. Spatial scales

Desertification can be analysed at different spatial scales, ranging from local to national and international levels. The perception of the scale and the seriousness of land degradation will be influenced by the timing of any investigation (Stocking and Murnaghan, 2001). For example: (a) many different forms of soil loss are most easily seen during or shortly after periods of heavy rains; (b) some types of erosion may be less visible after crops become established in fields; (c) and nutrient deficiencies and other factors that affect crop production will be best observed when crops are in field and relative growth rates can be assessed (Stocking and Murnaghan, 2001).

Some desertification indicators can only be understood at the national or international level. These are always indicators obtained from economic data, such as: arid and semiarid land percentage, areas and percentages of deforestation-reforestation, cultivated area per capita, food production per capita, and so on (ROSELT, 1997). Other indicators can be locally analyzed and must be determined over smaller space units, even when they can be aggregated at a national level. These are ecological and some of the economic indicators related, for instance, to production systems, the balance between production and consumption, or migrations (ROSELT, 1997). These indicators require a precise definition of the areas in which they are meaningful, and may not be applicable at larger mapping scales. Local indicators are derived from data collected in areas, which must be selected as representative of larger areas. At a national level, the collection of data destined for creating indicators must be based on a network of observatories that are representative in terms of ecology, climate and agriculture (ROSELT, 1997).

Different kinds of indicators may be relevant at different scales and meaningless at others. For example, as the technology of remote sensing and geographical information systems (GIS) advances, more and more indicators may be generated directly at the scale of interest. It is important to know which indicators are appropriate at different scales, and which of them can be obtained directly and in a cost-effective way at the desired scale.

#### 3.1.5.2. Temporal scales

Indicators need to be sensitive to changes in both management and climate. Soil characteristics that change within only a few weeks or months in response to the changing seasons, shifting weather patterns, and plant growth cycles are not appropriate soil quality/degradation indicators (Norton et al., 1999). Characteristics that begin to show change only after five or more years are not helpful indicators; often showing progressive soil degradation only after much of the productive topsoil is lost. The best soil quality/ degradation indicators are those characteristics that show significant change between one and three years, with five years being an upper limit of usefulness (Norton et al., 1999). Thus, the duration of an indicator's significance varies with the permanence of the data used to build the indicator. Some data change over the long term (e.g. topography and river networks) and are relatively permanent: others may change over the medium term (e.g. flora or erosion type); and some show short-term changes over years or seasons (e.g. soil moisture or livestock management) (ROSELT, 1997).

### 3.1.6. TYPES OF INDICATORS

Desertification indicators can be developed to define the desertification risk for a certain piece of land and for continued environmental monitoring. Such indicators can be divided into a number of different types: state, pressure, response, driving forces and impact indicators.

#### 3.1.6.1. State indicators

They provide an indication of the state of land degradation, or a particular aspect of it, at a given point in time. This pertains to qualitative and/or quantitative indicators (Fogh Mortensen, 1997). These kinds of indicators are related to the quality of the environment, they determine its status for a specific theme or field with reference to functions, space and time (ROSELT, 1997). The status is described by physical and natural environments (soil water availability, soil erosion vulnerability, land suitability to support specific type of land use) (Desertlinks, 2001). These are translated in terms of potentials, referring thus to notions of health (dysfunction, threatened species), abundance, diversity and distribution.

#### 3.1.6.2. Pressure indicators

These indicators are related to the constraints, damages and harmful effects endured by the environment through pressures of various origins (human, physical, etc.). Knowledge of the pressures on the environment is accompanied by an appraisal of the factors, which initiate them (human impact, natural causes, and so on). The pressures are mainly evaluated in physical terms (sometimes expressed as standards), but the evolution of costs for damage to the environment is another measurement (ROSELT, 1997). These indicators result from the driving forces, imposing unsustainable land-use practices and overexploitation of natural resources (deforestation, forest fires, ground water overexploitation) (Desertlinks, 2001).

#### 3.1.6.3. Response indicators

Response indicators provide a measure of the willingness and effectiveness of a society in providing responses, thus, these indicators are related to the corrective actions used to improve the status or reduce the pressures influencing the status. These actions are a result of private or government involvement. Indicators to describe them include an evaluation of the means to apply policies and the costs born by the producers, as well as the behavioral changes of people. They must provide measures of the action being taken to mitigate anthropic pressure and other factors that contribute to desertification process, to check the establishment of key components of monitoring system, the suitability of invested resources, the necessity of new laws, the integration of the land degradation problem in different levels policies (national, regional, local), the availability of financial resources to support targets, etc. They are expressed through preventive or corrective actions (implementation of laws and standards, intervention and management operations) (ROSELT, 1997). Thus, these indicators are related to implementation of programs for protecting areas from desertification, such as the application of sustainable farming systems, terracing, ground water recharge, storage of runoff water, controlled grazing, protection forests from fires) (Desertlinks, 2001). These indicators can be referred to four sectors of response:

- Capacity of immediate measures: incentives for area protection, ecosystem restoration, and so on.
- Capacity of structures and tools in the country: university, research centres, existence of databases, etc.
- Awareness: understanding of the meaning of desertification, presence of non governmental organizations, their number, size and age, etc.
- Funding.

A part from these three main categories of indicators listed above, the Desertlinks Project (2001), included two more categories: driving forces and impact indicators.

#### 3.1.6.4. Driving forces indicators

This kind of indicators represents human activities, processes and patterns that have an impact on land degradation. They provide an indication of the causes of positive and/or negative changes in the state of land degradation (Fogh Mortensen, 1997). Therefore, these indicators are related to the main human-induced causes of land degradation, such as: intensification of agriculture, overgrazing, increase of local population, and increase of tourists.

#### 3.1.6.5. Impact indicators

Indicators resulting from land degradation and desertification and related to on-site (loss in plant productivity, loss in farm income) or off-site impacts (flooding of low land, dam sedimentation). These are particularly important in that they provide information regarding the economic and social costs of desertification and the effects of drought.

## 3.2. IDENTIFICATION OF DESERTIFICATION INDICATORS

Over the past decades, scientists have worked on developing a set of basic soil characteristics that serve as key soil quality/degradation indicators (Doran and Parkin, 1996; Parr et al., 1992), but this work is difficult to put into practice because of the different interpretations, taking into account the numerous causes and effects of the land degradation processes and their interlinkages.

Indicators of land degradation must not only identify the condition of the soil resource but also define the economic and environmental sustainability of land management practices (Doran et al., 1999), although in the present work, only indicators related to the quality and health of the soil have been taken into account.

### 3.2.1. LAND-DEGRADATION INDICATORS

Soil degradation is one manifestation of land degradation that focuses on soil quality and soil productivity. Although soil degradation is only one aspect of land degradation, variables of its progress can be used as indicators of land degradation (Stocking and Murnaghan, 2001).

Soil quality/degradation indicators are usually classified as physical, chemical, and biological indicators (Doran and Parkin, 1996; Parr et al., 1992). Physical indicators include: soil texture, depth of soil, topsoil and rooting, bulk density, infiltration rate, and water retention characteristics. The chemical indicators are: total organic Carbon (C), Nitrogen (N), pH, electrical conductivity (EC), and extractable Nitrogen (EN), Phosphorus (P), and Potassium (K). The basic biological indicators are: microbial biomass C and N content, potentially mineralizable N, soil respiration, water content and soil temperature (Norton et al., 1999). The divisions between physical, chemical and biological factors are not clear-cut (Norton et al., 1999). Soil physical characteristics are affected by the biological and chemical condition of the soil. In turn, the soil biological activity is impacted by the soil's chemistry and physical structure. Likewise, the chemistry is impacted by other soil factors (Norton et al., 1999).

#### 3.2.1.1. Indicators of soil condition: state indicators

The condition of the soil may change considerable over a short time and may be considered as a time-dependent soil property. Soil condition is affected by antecedent moisture content, the length of time the soil is at a given moisture content prior to rainstorms, the wetting rate of the rain, etc. Soil condition is affected by the composition but is largely a function of the state that the materials comprising the soil are in at a particular time. Soil condition may vary depending on soil composition (texture, clay mineralogy), soil, pH, saturating cations, moisture content, organic content, etc (Norton et al., 1999). The condition of the soil is one of the best indicators of land degradation. The soil integrates a variety of important processes involving vegetation growth, overland flow of water, infiltration, and land use and land management. Soil degradation is, in itself, an indicator of land degradation. But, in the field, further variables are used as indicators of the occurrence of soil degradation (Stocking and Murnaghan, 2001). According to Doran et al. (1999); Larson and Pierce (1991) and Doran and Parkin (1994; 1996), the minimum data set of soil physical, chemical and biological indicators of soil condition are: (a) physical indicators: parent material, texture, colour, soil and plant rooting depth, soil structure, water holding capacity; (b) chemical indicators: soil organic matter content, pH, electrical conductivity, cation exchange capacity; and (c) biological indicators: soil respiration, soil temperature, microbial population.

#### 3.2.1.1.1. Main physical indicators of soil condition

##### 3.2.1.1.1.1. Parent material

Three of the most frequently observed soil indicators related to soil condition and land degradation are soil texture, colour and rooting depth. The three variables are intrinsically a function of the parent material of the soil and rate of weathering, and directly relate to production through the biophysical processes of plant growth in supplying nutrients and water and is providing a medium conducive to plant growth.

Table 3.1. Parent material classification based on its petrology and mineralogy

| Major class | Group | Type |
|---|---|---|
| Igneous rock | Acid igneous | Granite, grano-diorite, rhyolite, pyroclastics |
| | Basic igneous | Gabbro, basalt, dolerite |
| | Ultrabasic igneous | Peridotite, pyroxenite, ironstone, serpentine |
| Metamorphic rock | Acid metamorphic | Quartzite, gneiss, slate, phyllite |
| | Basic metamorphic | Schist, gneiss rich in ferro-magnesian, marble |
| Sedimentary rock | Clastic sediments | Conglomerate, sandstone, silstone, mudstone, claystone, shale limestone, marl |
| Unconsolidated | Clastic sediments | Fluvial, lacustrine, marine, colluvial |

The parent material may be defined using the geological map of the study area. The various types of parent materials are grouped into several classes according to their petrology and mineralogical composition (Kosmas et al., 1999) (Table 3.1). Likewise, according to Kosmas et al. (1999), the parent material may also be classified on the basis of its sensitivity to desertification (Table 3.2.).

Table 3.2. Parent material classification according to its sensitivity to desertification

| Class | Description | Parent material |
|---|---|---|
| 1 | Good | Shale, schist, basic, ultra basic, conglomerates, unconsolidated |
| 2 | Moderate | Limestone, marble, granite, rhyolite, ignibrite, gneiss, siltstone, sandstone |
| 3 | Poor | Marl, Pyroclastics |

### 3.2.1.1.1.2. Texture

Texture is dependent on the size and shape of particles and, therefore, on the mix of sand, silt and clay making up the soil. Soil texture is important for two reasons: first, (a) particle size and shape influence the likelihood of loss through wind or water, and second (b) soil texture also affects the infiltration rate of water, which in turn influences the amount of surface runoff and the potential for removal soil particles. For land degradation assessment, the texture is categorised into:

**(a) Sandy:** sand size particles predominate; low intrinsic fertility; easy to degrade; fine and medium sands susceptible to wind erosion. These soils tend to be more prone to drought than clayey soils because they retain less water at field capacity and the water retained is consumed more rapidly by the growing plants.

**(b) Loamy:** balanced proportions of sand, silt and clay, plus usually abundant organic matter; fertile; no major use limitations; difficult to degrade. It tends to have the largest available water holding capacity.

**(c) Clayey:** dominated by clays (either active clays or highly weathered stable clays); susceptible to several degradation processes such as waterlogging; high intrinsic fertility; variable sensitivity to degradation.

According to Stocking and Murnaghan (2001), soil particles may be classified according to their size (Table 3.3.)

Table 3.3. Classification of soil particles by size

| Description | Size | Symbol |
|---|---|---|
| Sand | 0.050- 2.000 mm | S |
| Silt | 0.002- 0.050 mm | Si |
| Clay | < 0.002 mm | C |

According to Kosmas et al. (1999), the soil textural classes can be grouped based on their water holding capacity (Table 3.4.).

Table 3.4. Soil textural class classification according to its water holding capacity

| Class | Description | Texture |
|---|---|---|
| 1 | Good | L, SCL, CL, SL, LS |
| 2 | Moderate | SC, SiL, SiCL |
| 3 | Poor | Si, C, SiC |
| 4 | Very poor | S |

Soil texture is related to erodibility, water-retention capacity, crusting, and aggregate stability (Kosmas et al., 1999). The amount of water available is related to both texture and soil structure. For land-degradation assessment through texture, it is important to select the least degraded soil and compare it with a degraded field soil. If water erosion has been prevalent, the loss of organic matter and selective removal of silt and clay will influence texture (Stocking and Murnaghan, 2001). Stones, varying from fragments of quartz to large pebbles, are also aspects of texture affected by degradation.

#### 3.2.1.1.1.3. Colour

Soil colour is constituted of the overall hue (based on primary colours), chroma (the strength of the colour) and the degree of greyness (from black to white) of the soil. When soil degradation takes place, both texture and colour change. Colour changes, especially in recently cultivated land, are often one of the first obvious indicators of land degradation.

Munsell soil-colour charts give a full description and code for soil colours. It is necessary to standardize the moisture level of the soil for the colour determination. For land-degradation assessment it is necessary to compare colours between undegraded and degraded conditions. Holding samples of the soil from the two conditions is an immediate indicator of soil degradation. The Munsell colour values give a semi-quantitative measure. At a larger scale in field, the occurrence of lighter patches in a field is often the result of topsoil loss and exhumantion of subsurface, which is naturally sandier and lighter in colour. Soil colour varies greatly with soil depth. Care needs to be exercised that only surface soil is used, so that differences in colour can then be ascribed to soil erosion.

#### 3.2.1.1.1.4. Soil and plant rooting depth

Soil depth is the vertical depth of soil from the surface down to weathered rock or other impermeable barrier, such as a stone-line or hardpan. It is the depth of the soil profile from the soil surface to the top of the regolith or unweathered parent material (Kosmas et al., 1999). Rooting depth describes the depth available to plant roots, but it is considered as the same as soil depth (Stocking and Nurmughan, 2001). In Table 3.5., soil depth is classified according to four different groups (Kosmas et al., 1999).

**Table 3.5. Soil depth classification**

| Class | Description | Depth (cm) |
|---|---|---|
| 1 | Deep | > 75 |
| 2 | Moderate | 75-30 |
| 3 | Shallow | 15-30 |
| 4 | Very shallow | < 15 |

The depth of soil material above weathered rock is a product of climate, which determines the rate of chemical breakdown of rocks, and the type of rock. Some rocks break down more quickly than others. The specific depth at any one site is determined by the balance between natural forces of removal of topsoil and the formation of new soil in the subsurface. The faster the rate of weathering and the more susceptible the rocks are to breakdown, the deeper the soil. Soil loss, through erosion by wind or water, serves to reduce the topsoil and thus the rooting depth. The other main way that soil depths and plant-rooting depths may be reduced is the formation of an impermeable layer induced by land use or agricultural practices. It may be from ploughing when the soil is too wet, which results in a compacted layer below the plough blade, or it may form because of chemical compaction in and around stone lines. Formation of an impermeable layer is a direct land-degradation process. The rooting zone is the main supplier of nutrients and water for plants. If the rooting depth is available to a plant is insufficient to allow that plant to put down sufficient roots, the plant will exhibit less vigorous growth and primary productivity is likely to be reduced. The depth of soil required by different plants varies, as does their ability to put down roots. On one hand, it also has to be taken into account that some soils are shallower than others even before land degradation, and in some cases barriers to the rooting depth occur naturally and not as a result of any degrading process, and on the other hand, that the effective plant rooting depth may be controlled by other factors, such as groundwater or very sandy layers with no nutrients. Therefore, a visual inspection of depth should include observation of root distribution and possible reasons for lack of roots in any layer. Soil and plant-rooting depth are, therefore, important indicators of erosion, because they may directly affect production output if depth is limiting.

**3.2.1.1.1.5. Soil structure**

Soil structure refers to the shape, size and degree of development of the aggregation, if any, of the primary soil particles into naturally or artificially formed structural units (McLaren and Cameron, 1996). Aggregates are groups of soil particles, also called peds. Their stability depends on a binding agent such as organic matter or lime, which cements the particles, and on the force that breaks them down, such as rainfall erosivity or the action of a plough (Stocking and Murnaghan, 2001). Good soil structure and aggregation, which are considered desirable traits of good soil, are often reduced with erosion because of a loss of organic matter. Reduction in organic matter leads to less stable aggregates and soil-crust formation in many soils, especially those with a large amount of silt and little organic matter at or near the soil surface. The crust in turn results in reduced infiltration and air permeability and increased water runoff. Bulk density often increases with erosion (Frye et al., 1982). Two indicators are considered for the assessment of soil structure: bulk density and infiltration rate.

**(a) Bulk Density (BD):** the bulk density is defined as the mass of soil divided by the volume occupied by soil, water and air (Stocking and Murnaghan, 2001). It is a measure of the weight of the soil per unit volume, usually given on an oven-dry (110ºC) basis (Campbell and Henshall, 1991). Variation in BD is attributable to the relative proportion and specific gravity of solid organic and inorganic particles and to the porosity of the soil.

**(b) Infiltration rate (I):** The rate at which water can enter the soil surface is called the infiltration rate. It is important, first, because it affects the rate at which a soil may recharge with water and, secondly, because it affects the likelihood of surface runoff and hence erosion occurring during heavy rain or irrigation (McLaren and Cameron, 1996). The infiltration rate is affected by many of the soil properties such as: texture, structure and pore characteristics, as a result of changes in the hydraulic conductivity. It is worth emphasizing that aggregate stability and the presence of swelling clays can have a large effect on infiltration. The surface soil is subject to raindrop impact as well as other factors, such as traffic and stock damage, which can reduce the infiltration rate of the soil raindrops can cause considerable impact on the surface soil aggregate and these can be easily break down if they are weak and unstable. Soils which have been heavily cultivated and are low in organic matter are particularly susceptible to aggregate breakdown and surface-crust formation. The presence of a layer of dead plant material, such as a mulch, helps to reduce raindrop impact and maintains a high infiltration rate.

#### 3.2.1.1.1.6. Water-holding capacity

Water holding capacity (WHC) is considered to be synonymous with the term available soil water, and it is defined as the amount of soil water available to plants, in other words, the portion of water in a soil that can be readily absorbed by plant roots. How much water a soil can hold is very important for plant growth. All of the water held by soil is not available for plant growth. Thus, WHC of a soil is a very important agronomic characteristic. Soils that can hold a lot of water support more plant growth and are less susceptible to leaching losses of nutrients and pesticides. This is true because a soil with a limited water-holding capacity (i.e. a sandy loam) reaches the saturation point much sooner than a soil with a higher water-holding capacity (i.e. a clay loam). After a soil is saturated with water, all of the excess water and some of the nutrients and pesticides that are in the soil solution are leached downward in the soil profile.

Soil-water-holding capacity is controlled primarily by the soil texture and the soil organic matter content. Soil texture is a reflection of the particle-size distribution of a soil. In general, the higher the percentage of silt and clay sized particles, the higher the water holding capacity. The small particles (clay and silt) have a much larger surface area than the larger sand particles. This large surface area allows the soil to hold a greater quantity of water. The amount of organic material in a soil also influences the water holding capacity. As the level of organic matter increases in a soil, the water-holding capacity also increases, due to the affinity of organic matter for water.

### 3.2.1.1.2. Main chemical indicators of soil condition

#### 3.2.1.1.2.1. Soil organic matter content

Soil organic matter (SOM) is a primary indicator of soil quality (Larson and Pierce, 1991; Doran and Parkin, 1994; Acton and Gregorich, 1995). SOM is considered the single most important indicator of soil quality and the soil organic carbon (SOC) the most important in the upper few centimetres of the soil (Larson and Pierce, 1991). SOM is found in varying amounts in mineral soils and is almost always most concentrated in the uppermost horizon (Birkeland, 1999). A wide spectrum of materials makes up the soil organic matter, which ranges from undecomposed plant and animal tissue to humus (Singer and Munn, 1987). The reason SOM is considered so important is that it impacts all five of the soil functions and is an important part of the physical, chemical and biological processes in soils. Kay and Angers (2000) highlighted the following effects of SOM on soil quality for these three processes: (a) physical effects or soil aggregation, erosion, drainage, tilth, aeration, water holding capacity, bulk density, evaporation and permeability; (b) chemical effects including exchange capacity; metal complexing; buffering capacity; supply and availability of N, P, S, and micronutrients; and adsorption of pesticides and other added chemicals; and (c) biological effects or activities of bacteria, fungi, actinomycetes, earthworms, roots and other microorganisms.

On the other hand, SOM is not homogeneous in the soil. It encompasses the living microbial biomass within the soil, as well as the dead plant matter and partially decayed and transformed plant and animal residues. SOM also has many beneficial properties. It serves as a plant nutrient storehouse, improves soil structure and water holding capacity, and reduces the toxicity of toxic substances within the soil. It supports a greater and more varied microbial population, favouring biological control of plant diseases, and aids in micronutrient element nutrition of plants and in the solubilization of plant nutrient elements from insoluble minerals. SOM has a high adsorptive or exchange capacity for plant nutrient elements, aiding in soil fertility. SOM is a critical soil property, along with textural characteristics, for determining soil erodibility (Wischemier and Smith, 1978; Wischmeier and Mannering, 1969). Where the soil organic matter content of a soil falls below 2 %, the soil is more prone to erosion, because soil aggregates are less strong and individual particles are more likely to be dislodged (Stocking and Murnaghan, 2001). Erodibility is a soil’s susceptibility to erosive forces such as wind and water (Soil Science Society of America, 1997). The primary role of SOM in reducing soil erodibility is by stabilizing the surface aggregates. This leads of a reduction in crust formation and surface sealing and increases water infiltration rates (Le Bissonnais, 1990).

**3.2.1.1.2.2. pH**

The pH is the standard measure of acidity and alkalinity. High pH indicates alkalinity, often from salts. Low pH indicates acidity, often from the loss of nutrient cations (Stocking and Murnaghan, 2001). Soil pHs have an extreme range of 2 to 11, but most soils pHs range from 5 to 9. The pH of acid soils is ordinarily between 4 and 7 (Birkeland, 1999). It is seldom below 4 unless free acids are present due to oxidation of sulfides. As an intensity factor of soil acidity, soil pH is a master indicator of the soil system and influences many of the chemical and biological processes occurring in soils, though it produces no evidence in the total soil acidity. Soil pH is dependent on the ionic content and concentration in both the soil solution and the exchangeable cation complex adsorbed to the surfaces of colloids. The pH is a good indicator of Al toxicity, which is generally the most limiting factor of crop production in acid soils (Norton et al., 1999). Sever problems are normally encountered when pH is below 5 because the most toxic species of $Al^{3+}$ predominates in solution. No serious problems exist at pH > 5.5, particularly in highly weathered soils. The pH also indicates the solubilities of most chemical elements in soils and their associated availability, deficiency, or toxicity to plant growth. At low pH, Al and Mn become more soluble and can be toxic to plants. Solubilities of most micronutrients except for Mo decrease as pH increases, and deficiency symptoms are often seen when pH is greater that 7. The degree of Al hydrolysis is controlled by soil pH, which further affects total soil acidity and the level of Al toxicity to plants. Soil pH also has a significant impact on soil microbiological activity. It has been shown that the rates of mineralization and nitrification increase with the increase of pH up to 7 (Alexander, 1980).

#### 3.2.1.1.2.3. Electrical conductivity

The soil electrical conductivity (EC), is defined as the ability of a soil to transmit or conduct an electrical current and is expressed in the units mS/m or it can also be reported as dS/m (Doerge, 2001). The usefulness of soil conductivity stems from the fact that sands have a low conductivity, silts have a medium conductivity and clays have a high conductivity. Consequently, conductivity (measured at low frequencies) correlates strongly to soil grain size and texture. In addition to its ability to identify variations in soil texture, electrical conductivity has proven to relate closely to other soil properties that often determine the productivity of a field, such as: (a) cation exchange capacity, which has a relationship with EC and clay content through the formula: CEC = 0.6 * (% clay) + 2.0 * (% organic matter) in milliequivalents; (b) the depth to claypan: the response of conductivity to the presence of clay has been used to accurately predict the depth of top soil over a clay layer; (c) the water holding capacity/drainage: areas of droughtness or excess moisture typically have distinct textural differences and these can be identified using electrical conductivity(soils in the middle range of conductivity, which are both medium textured and have medium water-holding capacity may be the most productive); (d) the soil organic matter: organic carbon accumulates in the poorly drained soils, which have higher clay contents; and (e) the salinity: an excess of dissolved salts in the soil is readily detected by electrical conductivity.

Therefore, soil electrical conductivity is a measurement that correlates to soil properties affecting crop productivity, including soil texture, CEC, drainage conditions, organic matter level, salinity, subsoil characteristics, topsoil depth, pH, and water-holding capacity.

#### 3.2.1.1.2.4. Cation-exchange capacity

Cation-exchange capacity (CEC) is considered as the process of ions held by electrostatic forces between the negative clay charge and positive ion charge. The most common nutrient cations are calcium, magnesium, potassium and sodium (Stocking and Murnaghan, 2001). CEC is largely determined by the total negative charge of soils, and all factors that alter it as well. The CEC varies in amount and origin with the soil material, and it also varies with the clay mineral, and the highest values are related to the organic matter. The variation with clay mineral is due to a combination of ionic substitution and its extent, the degree of hydration, and the number of exchange sites at the edges of particles (Birkeland, 1999). Because organic matter can have such a high CEC, the presence of a small amount of SOM can greatly affect the CEC of the soil. In contrast, non clay minerals and rock fragments have a negligible effect on the CEC. Thus, one can obtain a rough estimate of the CEC of the clay fraction, and hence of the possible clay minerals present, by knowing the CEC of the soil and the amount of clay present (Birkeland, 1999).

#### 3.2.1.1.3. Main biological indicators of soil condition

##### 3.2.1.1.3.1. Soil Respiration

The respiration of plant roots and of soil microorganisms results in a decrease in the level of $O_2$ and an increase in the level of $CO_2$ in the soil. Respiration rate can be measured as the amount of $CO_2$ evolved, or the amount of $O_2$ taken up, per unit mass, or volume of soil per unit time. The respiration rate of a soil may be increased by 40-100% by the presence of plant roots (McLaren and Cameron, 1996). Over 90% of soil respiration takes place within the topsoil. The rate of respiration may be greatly enhance by tillage since this increases the availability of $O_2$ to soil microorganisms. Other factors, which affect the rate of respiration, are temperature, water content and amount of readily decomposable organic matter in the soil (McLaren and Cameron, 1996). The respiration rate of plant roots and soil microorganism increases exponentially with temperature, within the normal range found in soils. Since soil temperature fluctuates with the time of year, or season, the respiration rate also varies with season.

#### 3.2.1.1.3.2. Soil Temperature

The temperature of the soil has a large effect on soil development and on land use. In cold soils both the rate of chemical weathering of soil minerals and the rate of biological cycling of nutrients are slow. The rate at which organic matter is decomposed by soil microorganisms increases with temperature between the range 5 to 30ºC or 40ºC (McLaren and Cameron, 1996). Plant growth is also affected by soil temperature and although the optimal temperature for growth depends on the species it is usually between 25 and 30ºC. The growth rate of plants and microorganisms can increase two or threefold for each 10ºC increase in soil temperature below their optimum temperature.

#### 3.2.1.1.3.3. Microbial population

The dynamic nature of soil biological communities, microbial and macrofaunal, makes them a sensitive indicator for assessing alterations in soil quality mainly due to changing management practices (Kennedy and Papendick, 1995). Soil populations may provide advanced evidence of subtle changes in the soil before changes in soil physical and chemical properties become apparent. Management practices on the land result in changes in soil physical and chemical properties, altering the soil environment that supports the growth of the microbial population.

## 3.2.2. INDICATORS RELATED TO LAND DEGRADATION PROCESSES: A QUANTITATIVE OR SEMI-QUANTITATIVE ASSESSMENT

### 3.2.2.1. Degradation of the vegetative cover

The quality of the vegetation is assessed in terms of: (a) plant cover, (b) drought resistance, (c) erosion protection to the soils, and (d) fire risk and ability to recover.

#### 3.2.2.1.1. Plant cover

Vegetation cover is defined in classes according to its relationship with soil erosion and land degradation. This indicator can be assessed measuring the decrease in percentage of canopy cover per year (%/ yr) (Poesen, 1995) (Table 3.6).

Table 3.6. Vegetation classification according to its cover percentage

| Class | Description | Plant cover (%) |
|---|---|---|
| 1 | High | > 40 |
| 2 | Low | 10-40 |
| 3 | Very low | < 10 |

#### 3.2.2.1.2. Drought resistance

Five categories are used for classification of vegetation with respect to drought resistance (Table 3.7).

Table 3.7. Vegetation classification according to its drought resistance

| Class | Description | Type of vegetation |
|---|---|---|
| 1 | Very high | Mixed macchia/ evergreen forests |
| 2 | High | Macchia, pine forest, permanent grasslands, evergreen perennial crops |
| 3 | Moderate | Deciduous forests |
| 4 | Low | Deciduous perennial agricultural crops (almonds, orchards, etc.) |
| 5 | Very low | Annual agricultural crops (cereals), annual grasslands, vines,… |

#### 3.2.2.1.3. Erosion protection to the soils

Five categories have been established according to the protection of the different types of vegetation to the soils in terms of risk of soil erosion and runoff generation.

Table 3.8. Vegetation classification according to its soil protection

| Class | Description | Type of vegetation |
|---|---|---|
| 1 | Very high | Mixed macchia/ evergreen forests |
| 2 | High | Macchia, pine forest, permanent grasslands, evergreen perennial crops |
| 3 | Moderate | Deciduous forests |
| 4 | Low | Deciduous perennial agricultural crops (almonds, orchards, etc.) |
| 5 | Very low | Annual agricultural crops (cereals), annual grasslands, vines,… |

#### 3.2.2.1.4. Fire Risk and ability to recover

The different types of vegetation may be grouped into for example four categories according to the fire risk (Table 3.9). It has to be taken into account that the vegetation categories may differ according to the area of study.

Table 3.9. Vegetation classification according to the risk of fire

| Class | Description | Type of vegetation |
|---|---|---|
| 1 | Low | Bare land, perennial agricultural crops, annual agricultural crops (maize, sunflower, tobacco, etc.) |
| 2 | Moderate | Annual agricultural crops (cereals, grassland), deciduous oak, macchia/ evergreen forests |
| 3 | High | Macchia |
| 4 | Very high | Pine forest |

### 3.2.2.2. Soil-degradation processes

#### 3.2.2.2.1. Physical soil degradation (The analysis refers to the 60 cm of the soil layer)

##### 3.2.2.2.1.1. Compaction and crusting

Bulk density (BD), is identified as one indicator of the level of compaction and crusting of the soil. Its rate can be expressed by the increase in the BD in cm/h/yr (Poesen, 1995; Balba, 1995). According to FAO/UNEP/UNESCO (1979), four degradation classes can be distinguished (Table 3.10).

Table 3.10. Soil physical degradation classes based on the increase in bulk density (% change/yr) with reference to initial level (g/cm3)

| | **Initial level in g/ cm3** | | | |
|---|---|---|---|---|
| Degradation classes | 1.0 | 1.0- 1.25 | 1.25- 1.4 | 1.4- 1.6 |
| None to slight | < 5.0 | < 2.5 | < 1.5 | < 1.0 |
| Moderate | 5.0- 10 | 2.5- 5.0 | 1.5- 2.5 | 1.0- 2.0 |
| High | 10-15 | 5.0- 7.5 | 2.5- 5.0 | 2.0- 3.0 |
| Very high | > 15 | > 7.5 | > 5.0 | > 3.0 |

A second indicator of the compaction and crusting of the soil is infiltration rate (I). Its rate can be expressed by the decrease in the I, decrease in soil permeability (P) in cm/h/yr (Poesen, 1995; Balba, 1995). According to FAO/UNEP/UNESCO (1979), four degradation classes can be distinguished (Table 3.11).

Table 3.11. Soil physical degradation based on the decrease in I (% change/yr) with reference to initial level (cm/hr)

| | **Initial level in cm/hr** | | |
|---|---|---|---|
| Degradation classes | Rapid: 20 | Moderate: 5- 10 | Slow: 5 |
| None to slight | 2.5 | 1.25 | 1.0 |
| Moderate | 2.5- 10 | 1.25- 5.0 | 1.0- 2.0 |
| High | 10- 50 | 5.0- 20 | 2.0- 10 |
| Very high | >50 | >20 | >10 |

#### 3.2.2.2.2. Chemical soil degradation

Both acidification and increase in toxic elements are often active mainly in the topsoil. Thus, the soil layer to 30 cm depth is referred to them, in order not to dilute the effect by taking another 30 cm of possibly little affected soil into account (Balba, 1995).

##### 3.2.2.2.2.1. Increase in toxic elements

An indicator of the increase in toxic elements may be the concentration of each toxic element, in terms of increase of concentration of a determined pollutant in mg/Kg.

3.2.2.2.2.2.

### 3.2.2.2.2.3. Acidification

As above, the first way of determining the acidification of the soil is by direct measure. In addition, the base saturation (V) of the soil is also a good indicator of acidification. It is expressed as a decrease of V in percentage per year (%/yr):

V= [(Total exchangeable bases)/ (Cation exchange capacity)]/ 100 (3-1)

According to FAO/UNEP/UNESCO (1979), there can be different degradation classes distinguished according to the percentage of base saturation.

Table 3.12. Chemical degradation classes when the base saturation is less than 50%

| Degradation classes | Decrease in base saturation in %/ yr |
|---|---|
| None to slight | < 1.25 |
| Moderate | 1.25- 2.5 |
| High | 2.5- 5.0 |
| Very high | < 50 |

Table 3.13. Chemical degradation classes when the base saturation is more than 50%

| Degradation classes | Decrease in base saturation in %/yr |
|---|---|
| None to slight | < 2.25 |
| Moderate | 2.25- 5.0 |
| High | 5.0- 10 |
| Very high | > 10 |

### 3.2.2.2.2.4. Salinisation

Salinisation is referred here to the 60 cm depth of the soil layer. The electrical conductivity (EC) is one indicator of the salinisation of the soil. Its rate can be expressed by the increase in EC of saturated paste extract at 25ºC, in dS/m (Balba, 1995) or in mmhos/cm/yr (Poesen, 1995). The degradation classes (FAO/UNEP/UNESCO, 1979) are presented in Table 3.14.

Table 3.14. Chemical degradation classes based on the soil electrical conductivity

| Degradation classes | Salinization: increase in EC in dS/m/yr |
|---|---|
| None to slight | 2 |
| Moderate | 2-3 |
| High | 3-5 |
| Very high | 5 |

#### 3.2.2.2.2.5. Sodification

Sodification is referred here to the 60 cm depth of the soil layer. The exchangeable sodium percentage (ESP), is one indicator of the level of sodification of a soil. It is expressed as the increase of ESP in percent per year (%/yr) (Poesen, 1995; Balba, 1995).

$$ESP= [(\text{Exchangeable sodium})/ (\text{Cation exchange capacity})] \times 100 \qquad (3\text{-}2)$$

According to FAO/UNEP/UNESCO (1979), four different degradation classes are presented according to the sodium percentage (Table 3.15).

Table 3. 15. Chemical degradation according to the exchangeable sodium percentage

| Degradation classes | Sodification: increase in ESP in %/yr |
|---|---|
| None to slight | 1 |
| Moderate | 1-2 |
| High | 2-3 |
| Very high | 3 |

#### 3.2.2.2.3. Biological soil degradation

As biological degradation is also very much a topsoil phenomenon, the 30 cm depth is referred to (Balba, 1995).

#### 3.2.2.2.3.1. Loss of organic matter/ humus

Soil organic matter (SOM) or Humus content is considered as the main indicator of the loss of soil organic matter. It is measured as the decrease in SOM in percentage per year (%/yr) (Table 3.16).

Table 3.16. Chemical degradation classes due to the decrease of soil organic matter

| Degradation classes | Decrease in SOM or Humus in %/yr |
|---|---|
| None to slight | < 1.0 |
| Moderate | 1.0- 2.5 |
| High | 2.5- 5.0 |
| Very high | > 5.0 |

### 3.2.2.2.4. Soil erosion

#### 3.2.2.2.4.1. Water erosion

The main indicator of water erosion is the loss of soil. The rate or intensity of soil erosion is usually expressed as soil loss per soil surface unit and per interval the time (Poesen, 1995), such as: t/ha/yr. It can also be expressed in mm/yr (Balba, 1995). The main degradation classes (FAO/UNEP/UNESCO, 1979) are illustrated in Table 3.17.

Table 3.17. Soil degradation classes by water erosion

| Degradation classes | Soil loss in: t/ ha/ yr | Soil loss in: mm/ yr |
|---|---|---|
| None to slight | 10 | 0.6 |
| Moderate | 10-50 | 0.6-3.3 |
| High | 50-200 | 3.3-13.3 |
| Very high | 200 | 13.3 |

#### 3.2.2.2.4.2.Wind erosion

The main indicator of wind erosion is the loss of soil by the action of wind (in t/ha/yr or in mm/yr).

According to FAO/UNEP/UNESCO (1979), the degradation classes are grouped according to soil loss (Table 3.18).

Table 3.18. Soil degradation classes by wind erosion

| Degradation classes | Soil loss in: t/ ha/ yr | Soil loss in: mm/ yr |
|---|---|---|
| None to slight | 10 | 0.6 |
| Moderate | 10-50 | 0.6-3.3 |
| High | 50-200 | 3.3-13.3 |
| Very high | 200 | 13.3 |

## 3.2.3. MAIN INDICATORS OF LAND DEGRADATION BASED ON FIELD DETERMINATION

Visual indicators of land degradation can be derived from visual and morphological observations in the field. These kinds of indicators are designed to be quick to measure, and the recommendation is to do as many as possible measures of each one (minimum of 20 observations) (Stocking and Murnaghan, 2001).

### 3.2.3.1. Soil degradation

#### 3.2.3.1.1. Physical soil degradation

Physical degradation is constituted as a set of types of soil degradation involving one or more of the following processes: loss of soil physical structure, sealing and crusting of soil surface; reduction in permeability; compaction at depth; increase in macroporosity and limitations to rooting (Stocking and Murnaghan, 2001). Some other specific indicators of physical soil degradation that may be determined in the field are:

(a) Increased incidence of plant disease/ morphological irregularities (i.e. stunting)
(b) Decreasing crop yields
(c) Plough pan
(a) Restricted rooting depth

(b) Structural degradation, including compaction

(c) Increased sealing, crusting and runoff; reduced soil water

#### 3.2.3.1.2. Chemical soil degradation

This type of land degradation may involve one or more of the following processes: leaching of nutritive elements, acidification, toxicities, and excess of salts (Stocking and Murnaghan, 2001). Some specific indicators of chemical soil degradation that may be determined in the field are:

(a) Low pH

(b) Nutrient deficiency/ toxicity symptoms evident on plants

(c) Increased incidence of plant disease/ morphological irregularities (i.e. stunting)

(d) Decreasing crop yields

(e) Changes in vegetation species

(f) Poor response to fertilizers

(d) Increased sealing, crusting and runoff; reduced soil water

*Salinity or alkalinity:*

Salinity is considered a type of soil degradation where salts increase in the soil water solution. Alkalinity is defined as a type of soil degradation where sodium cations increase on the exchange complex of clay and organic matter particles in the soil. Increased alkalinity leads to physical degradation as well as chemical problems (Stocking and Murnaghan, 2001). Specific indicators of soil salinity or alkalinity are:

(a) Efflorescence

(b) Soil particles unstable in water

(c) High pH

(d) Increased incidence of plant disease/ morphological irregularities (i.e. stunting)

(e) Decreasing crop yields

(f) Changes in vegetation species

(g) Structural degradation, including compaction

(e) Decrease in organic matter (lighter-coloured soils)

(f) Increased sealing, crusting and runoff; reduced soil water

#### 3.2.3.1.3. Biological soil degradation

It is a type of soil degradation consisting of the mineralization of humus and an increase in the activity of microorganisms responsible for organic decay, resulting in an overall decrease in organic matter content (Stocking and Murnaghan, 2001). Some specific indicators of biological soil degradation that may be determined in the field are:

(a) Low pH

(b) Nutrient deficiency/ toxicity symptoms evident on plants

(c) Decreasing crop yields

(d) Poor response to fertilizers

(e) Decrease in organic matter (lighter-coloured soils)

(f) Increased sealing, crusting and runoff; reduced soil water

(g) Decrease in number of earthworms/ ants and similar

### 3.2.3.2. Soil erosion

#### 3.2.3.2.1. Water erosion

Specific indicators of soil erosion by water that may be determined in the field are:

(a) Rills

(b) Gullies

(c) Pedestals

(d) Armour layer

(e) Accumulations of soil around clumps of vegetation or upslope of trees, fences or other barriers

(f) Deposits of soil on gentle slope

(g) Exposed roots or parent material

(h) Sedimentation in steams and reservoirs

(i) Low pH

(g) Nutrient deficiency/ toxicity symptoms evident on plants

(h) Decreasing crop yields

(i) Changes in vegetation species

(m) Restricted rooting depth

(n) Decrease in organic matter (lighter-coloured soils)

(o) Increased sealing, crusting and runoff; reduced soil water

#### 3.2.3.2.2. Wind erosion

Specific indicators of soil erosion by wind that may be determined in the field are:

(a) Pedestals

(b) Armour layer

(c) Accumulations of soil around clumps of vegetation or upslope of trees, fences or other barriers

(d) Exposed roots or parent material

(e) Dust storms/ clouds

(f) Sandy layer on soil surface

(g) Decreasing crop yields

## 3.3. DEVELOPMENT OF COMPOSITE INDICATORS

### 3.3.1. DEFINITION AND CONCEPT OF COMPOSITE INDICATORS

Single indictors give singular items of evidence for land degradation or its impact, but they are susceptible to error, misinterpretation and chance (Stocking and Murnaghan, 2001). The use of only one indicator to conclude definitively that land degradation has occurred or it is occurring is problematic, and often imprecise and inaccurate. While each single indicator has its own attributes and applications, several indicators combined can piece together a far more comprehensive and consistent understanding of the problem. Therefore, combining indicators, more robust conclusions can be achieved, even to the extent that quite different types of measure may be placed alongside each other to obtain a fuller understanding of whether land degradation is taking place or not (Stocking and Murnaghan, 2001). According to the Desertlink Project (2001), composite indicators are considered different from individual indicators in two ways:

(a) They have been derived from a scientific understanding of the processes of desertification based on theoretical and empirical research over the last decade.

(b) They are a direct consequence of our scientific understanding; they provide two kinds of composite indicators for desertification risk, combining simple factors to reflect their interactions and the range of conditions under which each is important.

Stocking and Murnaghan (2001), highlighted three different areas of combining indicators:

(a) Combinations to show both the process and likely cause of land degradation through time.
(b) Combinations to provide corroborating evidence and a consistent view of land degradation.
(c) Methods to bring individual indicators together for comparative and overall assessment, including how to search for a suite of indicators and how to develop a semi-quantitative procedure for getting an overall assessment of the problem.

### 3.3.2. GUIDELINES FOR COMBINING INDICATORS

It is difficult to provide specific guidance for all land-degradation situations, due to the many possible permutations of possible land degradation and land use conditions, and hence many possible interpretations. According to Stocking and Murnaghan (2001), there are two different steps in assessing land degradation: the approach to adopt in the field and how to put the indicators together in a semi-quantitative form for initial inspection.

### 3.3.2.1. General guidance for field assessment

It is important to make a careful reconnaissance of the field site to note all the pieces of evidence of both land-degradation processes and their impacts (Stocking and Murnaghan, 2001).

(a) Map out the field slope as a sketch, noting the position of any obvious features such as gullies, rills, tree mounds, and so on. Any differential crop growth or obvious nutrient deficiencies should also be noted on the map.

(b) Obtain the history of land use: when the plot of land started to be used, crops grown, any change in land use, subdivisions of the land, and similar important events that could have a bearing on land degradation. These events should later be set alongside the field measurements to ascertain whether they correspond with observations.

(c) Determine any significant events: wildfires, exceptionally heavy storms and soil wash, dates when trees were cut down, and so on.

(d) Note any particular farming techniques that may have implications for land degradation.

(e) Identify the indicators of the processes of land degradation:

- Soil losses from single places, i.e. tree mounds, pedestals, soil depth.
- Soil losses from small parts of the field, i.e. rills, armour layer.
- Soil losses from large parts of the whole of the field, i.e. gullies; differences in soil depth between degraded field and non-degraded; or averages over the field of previous items such as tree mounds.

- Sediment accumulations and their enrichment/texture within the field, i.e. drainage ditches
- Sediment accumulations and their enrichment/texture at the base of the field, i.e. boundary accumulations.
- Sediment accumulations and their enrichment/texture outside the field, i.e. clay enrichment in hollows.

(f) Identify the indicators of the impact of land degradation:

- Observation of current plant growth, i.e. within-field differences.
- Actual measurements of different sizes of plants.
- List known nutrient deficiencies observed.
- Estimate likely yields from different parts of the field.
- Obtain historical yield, and observations on how plant growth has changed.

(g) Compile a comprehensive table of indicators and results, looking for trends, consistency and areas where there is broad agreement in the scale of degradation.

### 3.3.2.2. Semi-quantitative assessment

This kind of assessment is not an actual measure of land degradation, but a production of potential land degradation according to the environmental factors that encourage it. Such semi-quantitative techniques, therefore, provide both a current view and a predictive means to monitor land degradation status (Stocking and Murnaghan, 2001). Douglas (1997) suggested the development of simple scoring techniques for seriousness of indicators of land degradation, but it is perfectly appropriate to develop one's own scoring system. Provided that it is consistently used, it can be a good way of combining indicators to get a more comprehensive view of land degradation (Stocking and Murnaghan, 2001). The following two tables give examples of the combination of observations of a number of separate indicators:

Table. 3.19. Semi-quantitative assessment of the interrill erosion process

| Ranking | Degree | Description |
|---|---|---|
| X | Not apparent | No obvious signs of interrill erosion, but evidence of minor interrill erosion may have been masked, for instance by tillage. |
| 0 | No sheet erosion | No visual indicators of interrill erosion |
| 1 | Slight | Some visual evidence of the movement of topsoil particles downslope through surface wash; no evidence of pedestal development; only a few superficial roots exposed |
| 2 | Moderate | Clear signs of transportation and deposition of topsoil particles downslope through surface wash; some pedestalling but individual pedestals no more than 50 mm high; some tree and crop roots exposed within the topsoil; evidence of topsoil removal but no subsoil horizons exposed. |
| 3 | Severe | Clear evidence of the wholesale transportation and deposition of topsoil particles downslope through surface wash; individual pedestals over 50 mm high; extensive exposure of tree and crop roots; subsoil horizons exposed at or close to the soil surface. |

Table. 3.20. Semi-quantitative assessment of the rill erosion process

| Ranking | Degree | Description |
|---|---|---|
| X | No erosion | No rills present within the field |
| 1 | Slight | A few shallow (<100mm depth) rills affecting <= than 5% of the surface area. |
| 2 | Moderate | Presence of shallow to moderately deep rills (< 200 mm depth) and/ or rills affecting up to 25% of the surface area. |
| 3 | Severe | Presence of deep rills (up to 300 mm depth) and/or rills affecting more than 25% of the surface area. |

Such tables should be adapted for the specific circumstances of each field and the different types of land use. Once developed for a local area, they can provide excellent ratings to determine the specific danger of different types of land use in terms of land degradation. These kinds of assessments are very useful because they are easy to develop and interpret.

### 3.3.3. OUTCOMES OF LAND DEGRADATION AND DESERTIFICATION

By combining indicators of both the process of degradation and its impact, scenario outcomes can be developed. The outcomes of land degradation involve many factors; only some of which are directly measurable, but even these require repeated measurements over many years in order to construct reliable relationships with land degradation (Stocking and Murnaghan, 2001).

## 3.4. INDICATORS, GIS TECHNIQUES AND SPATIAL MODELLING: AN INTEGRATED APPROACH

### 3.4.1. INTER-LINKED SETS OF INDICATORS

In its most general sense, an indicator is a sign, which is defined as something that stands for something to somebody in some respect or capacity. At a more concrete level, an indicator is a variable, an operational representation of an attribute (quality, characteristic, property) of a system (Gallopin, 1996). Each variable is associated with a particular set of entities through which it manifests itself, and these entities are usually referred to as states (or values) of the variable. Thus, indicators are variables; data are actual measurements (quantitative) or observations (qualitative) of the values of the variables at different times, locations, populations, or combinations of these. A collection of quantitative data is usually referred to as statistics. At a given level of aggregation or perception (such as local or global), indicators can be defined as individual variables or as variables that are a function of other variables. The function may be from simple to very complex such as:

**(a) A ratio:** Including the concept of index number, measuring the change in the values of a variable relative to some base value.

**(b) An index:** A single number, which is a simple function of two or more variables, usually a weighted summation of individual variables, an averaging, a multiplication, or a maximum operation. An index is a simple function of lower-level variables (subindices).

**(c) Simulation model:** The aggregation of indicators into a single index, often as simple as a summation or averaging, is useful for assessing progress, but usually not for understanding early warning and forecasting. For this latter purpose it is necessary to choose indicators having some correspondence with meaningful attributes of the object studied, and to insure that some minimum correspondence exists between the functional inter-linkages among the attributes and the procedure linking or aggregating the individual indicators. A set of indicators (variables) and a set of assumed relations among them constitute a *model* of the original system. This model may be only a blurry mental image about how indicators are interconnected causally, or it may be a highly formalized mathematical model (Gallopin, 1997). Modelling can be a useful tool for understanding the inter-linkages between various indicators. The work on modelling may help building aggregated indicators or indices. Developing such indices reduces the number of indicators and adds analytical and interpretative value to the process, and it also, however, increases the value content (Waller-Hunter, 1996)

There is a need to move beyond the usual, more or less exhaustive, lists of individual indicators to integrated or interlinked sets of indicators. This is particularly important regarding the uses of indicators for early warning and for forecasting (Gallopin, 1997). These inter-linkages may be physical flows of matter or energy or causal influences that are not adequately described as material or energy flows. Linkages may exist among variables within a subsystem, among different subsystems, and among whole systems; and they can be referred to as "horizontal" linkages (Gallopin, 1997). "Vertical" or interlevel linkages, on the other hand, are those existing between the systems belonging to different levels of organization, such as: local, regional, global; or between subsystems belonging to different levels of the system, if the latter is defined as a hierarchical system (Gallopin, 1997).

## 3.4.2. USE OF MODELS TO LINK INDICATORS OF DESERTIFICATION AND LAND DEGRADATION

### 3.4.2.1. Correlation Models

One approach that has been used with some success to reduce the number of variables that a modeller or a decision-maker has to deal with is to compute correlation coefficients between all of the candidate variables (Rutherford, 1997). Those that are uncorrelated below a certain threshold are discarded as irrelevant and those that are nearly perfectly correlated are discarded as redundant. Correlation models can tell us something about the "linkages" between a larger number of variables, but they tell us nothing about why certain variables are correlated, and they deal only with linear correlation at that. Correlation results can stimulate research into new forms of conceptual models or complex system models.

### 3.4.2.2. Input-Output Models

Input-output models depend on the transfer function, but may be essentially linear and although they can handle a large number of variables of different types, they cannot deal with the feedback effects so characteristic of real, non-linear, systems. They are fully deterministic. Although they can and have been used to construct scenarios for various levels of inputs and outputs in a given range, may not be valid for many different ranges because of the absence of feedback and other non-linear effects that cannot be captured in such a model (Rutherford, 1997).

### 3.4.2.3. Complex System Models

Complex model systems are based on equations reflecting known relationships between the variables. They can fully non-linear, allow feedback, and demonstrate the full range of phenomena such as self-organizing structures, multiple quasi-stable states (attractors), sudden flips from one state to another, etc (Rutherford, 1997).

### 3.4.3. UTILITY OF THE GIS TECHNIQUES

#### 3.4.3.1. Definition of a Geographical Information System (GIS)

According to Calkins and Tomlinson (1977), a geographic information system is defined as "an integrated software package specifically designed for use with geographic data that performs a comprehensive range of data-handling tasks. These tasks include data input, storage, retrieval and output, in addition to a wide variety of descriptive and analytical processes". It is considered a mapping system with the capability to record, store, process, manipulate and retrieve data, in order to: (a) digitize maps, (b) overlay spatial data, (c) display information (needed in decision-making), and (d) manipulate attribute information from geographically referenced spatial data. GIS software can be either Raster (cell-based) or Vector-based systems.

#### 3.4.3.2. Advantages of the use of GIS

In this framework, one of the most relevant advantages of using geographical information systems (GIS), is that their technology allow the user to link with simulation models to create "what if" scenarios and compare observational data with these model-generated situations. Moreover, there are some other positive benefits of their use (Peuquet et al., 1993), such as: (a) GIS allow finer-grainer and more complex types of analysis, as well as a broader range of analytical and manipulative capabilities, than ever possible through manual methods; (b) they provide analytical results far more quickly and with mechanical accuracy; (c) GIS provide a fast and compact method of data storage; (d) data can be maintained and extracted at a lower cost per unit of data handled; (e) GIS impose a uniformity of procedures that can be easily repeated. There is a resulting impetus to integrate data collections, spatial analysis, and decision-making processes into a common information flow. This allows accessibility to a broader range of data while also fighting the problem of inconsistent data; and (f) that GIS have the ability to perform visual and mathematical overlays. Visual analysis could be quantified and used to examine causes, effects and interactions under different development scenarios. Therefore, GIS are a useful, and perhaps necessary, tool for incorporating land-degradation indicators into the development process. The integration of economic, social and desertification indicators in a spatial framework allows for more powerful and realistic analyses than those offered by conventional non-spatial methods (Winograd and Eade, 1997).

Geographical information systems (GIS), have been applied to software and hardware, particularly suited to deal with spatial data and information, and recently have been integrated with other types of information systems and tools (Database Management Systems, traditional office spreadsheet packages, and Internet and components). There are several benefits to using GIS in indicator work. According to Langaas (1997), some of the important benefits of this are: sampling, analysis, database management, visualization and GIS-WWW interlinkages.

#### 3.4.3.2.1. Sampling

In many cases, when dealing with environmental indicators, there is a need to obtain a spatially representative sample of an indicator in question, possibly for estimation of an average value for the overall area considered, assuming that the processing is linear. By offering geostatistical analytical tools, GIS can assist in making spatially unbiased averages from geographically distributed sample measurements.

#### 3.4.3.2.2. Analysis

One of the key features of GIS is the range of tools offered for spatial analysis, and frequently also for non-spatial analysis of data associated with the geographic features, called attributed data. As an example, the *Land affected by desertification* study, in (chapter 12 in UN/CSD 1996), proposed that the creation of an index, which combines degrees of severity, will require the following measures:

(a) Area subjected to severe land degradation: x_ $km^2$ (severe here includes both the severe and very severe categories of UNEP)
(b) Area subjected to moderate land degradation: y_ $km^2$.
(c) Area subjected to slight land degradation: z_ $km^2$.
(d) National area (excluding surface water bodies): n_ $km^2$.
(e) Natural area of drylands (vulnerable to desertification, assuming that all drylands are potentially vulnerable to desertification. Hyper-arid lands are excluded), consisting in arid, semi-arid, and sub-humid land: d_ $km^2$.

From the above measurements, the following sets of numbers can be derived:

a. National area affected by desertification = $(x + y + z)$ $km^2$
b. Per cent of national area affected by desertification = $[(x + y + z)/n]$ 100%
c. Percentages of national area affected by severe, moderate and slight desertification respectively can be calculated the same way.
d. Per cent of national drylands affected by desertification = $[(x + y + z)/d]$ 100%
e. National area not affected by desertification = $[n - (x + y + z)]$ $km^2$.
f. National dryland area not affected by desertification = $[d - (x + y + z)]$ $km^2$.

The development of these indicators clearly requires analysis of GIS data using Boolean overlay techniques, and can only be obtained with difficulty otherwise.

#### 3.4.3.2.3. Database management

Many of the more comprehensive GIS software packages are connected with powerful database management systems (DBMs). Thus, the use of DBMs found in GIS may be considered to keep and maintain Indicator Desertification System database, including the raw statistical data, and thereby improve the opportunities for cartographic visualization.

#### 3.4.3.2.4. Visualization

The production of cartographic outputs is a feature of all GIS. In an Indicator Framework, at least two types of cartographic outputs may be useful to produce spatial indicators maps and “reference” maps. Other types of cartographic illustrations in the context of reporting and visualization are reference or index maps, showing the locations of measurements stations (when a few measurement stations represent a larger region).

#### 3.4.3.2.5. GIS-WWW Interlink

GIS software is being directly connected to Internet server software (Langaas, 1997; Steinke et al., 1996), allowing the Internet user to individually define and visualize spatial indicators better suiting the user’s needs. On-line interactivity is not restricted to GIS-www connected databases, but also for DBMs-www ones, thus allowing for the individual definition and creation of graphs and charts (Langaas, 1997).

### 3.4.3.3. The Digital Elevation Model (DEM)

Digital Elevation Models are spatial fields of terrain elevation values that are usually arranged in a regular square grid or in other arrangements such as a Triangulated Irregular Network (TIN). There are a number of ways in which DEMs can be derived, including digitising contour lines from topographic maps, ground surveys using theodolites of levels, and stereo interpolation of pairs of aerial photographs, all of which contain measurement errors in position and elevation (Blöschl and Grayson, 2001).

#### 3.4.3.3.1. General information depicted from a DEM

(a) Slope: displays the grade of steepness expressed in degrees or as percent slope. This image can reveal structural lineaments, fault scarps, fluvial terrace scarps and other morphological features.

(b) Aspect: identifies the downslope direction. Aspect images may assist in the identification of landforms such as fluvial networks, alluvial fans and faceted fault related scarps.

(c) Shaded topographic relief or hill-shading: this image depicts relief by simulating the effect of the Sun's illumination on the terrain. The direction and the altitude of the illumination can be changed in order to emphasize faults and lineaments. This image is probably the most useful to display geological data related to landforms in terrains that show a close correlation between geology and topography.

(d) Flow direction: shows the direction of flow by finding the direction of the steepest descent or maximum drop. This DEM-derived surface depicts the drainage pattern.

(e) Basin: function that uses a grid of flow direction (output of flow direction) to determine the contributing area.

### 3.4.4. GEOGRAPHICAL INFORMATION SYSTEMS AND SPATIAL MODELLING

Nowadays, Geographical Information System (GIS) represents a powerful and flexible tool for managing resources and understanding and predicting complex and changing systems (Peuquet et al., 1993).

GIS need to incorporate a modelling component in order to predict potential outcomes and evaluate alternatives. Otherwise, they are restricted to displaying existing data and their value to environmental managers is limited. Current spatial modelling capabilities in GIS are generally limited to such two operations as:

(a) Map overlaying: it is the automated version of the cartographic process of detecting spatial correlations between two variables by overlaying maps showing their distributions.

(b) Adjacent analysis: it is also called buffering, and consists of drawing boundaries at a specified distance around a point, linear or areal feature on a map.

But only the simplest of land-degradation processes can be analysed with overlay and buffering operations. The usual solution to modelling with GIS is to provide GIS with links to external models where the operations can be performed in languages (FORTRAN, C, etc.) that are much better suited to complex mathematical calculations (Peuquet et al., 1993). Thus one can call an external model to calculate a parameter value and store that value in a database where it is accessible to the GIS. However, this solution makes neither the spatial data handling and display capabilities of the GIS accessible to the model, nor the predictions generated by the model easily available to the GIS database. Nevertheless, integration via data files is still the dominant method of connecting models to GIS.

## 3.5. SUMMARY

Land degradation is difficult to comprehend in its totality and indicators are considered an appropriate tool for its assessment. An effective indicator has to be quantifiable, relevant, understandable, reliable and cost effective; and its data must be either already available or they should be easily obtainable. Moreover, it is important to know which indicators are appropriate at different spatial scales, since some indicators might be meaningful at small scales but may not be applicable at larger scales, and vice versa (ROSELT, 1997). Regarding temporal scale constraints, the significance of an indicator varies with the permanence of the data used to build the indicator. Thus, the interval of change of a particular variable would be decisive for its identification as a proper indicator. The best soil quality/degradation indicators are those characteristics that show significant change between one and three years, with five years being the upper limit of usefulness (Norton et al., 1999).

This research will mainly focus on the indicators of soil condition as well as on the indicators related to land-degradation processes. The former type comprises physical, chemical and biological indicators, whereas the latter is based on the degradation of the vegetative cover, physical, chemical and biological degradation processes and soil erosion. In the development of the DIS model, indicators of land degradation and desertification will be combined with modelling tools and GIS techniques.

# 4. METHODOLOGY



## 4.1. INTRODUCTION

This chapter seeks to explain how the main aim of this thesis: the development of a desertification indicator system (DIS) for a semi-arid Mediterranean environment will be accomplished. Accordingly, it is structured in relation to the main specific objectives detailed in the chapter one. Therefore, two sections are described: (a) the assessment of the most important factors of land degradation at plot scale; and (b) the development and application of the DIS model. The interlinkages among these main sections are illustrated in the form of flow diagram in the Figure 4.1.

The first section describes the conducted research strategy, as well as the associated field determinations and laboratory analyses. This section has five main purposes: (a) the study of the global processes and causes of land degradation and the specific ones for the target area, (b) the description of the research strategy, (c) the assessment of the role of land use-cover on the main soil physico-chemical properties, (d) the assessment of land degradation processes, and (e) the development of indices of soil quality.

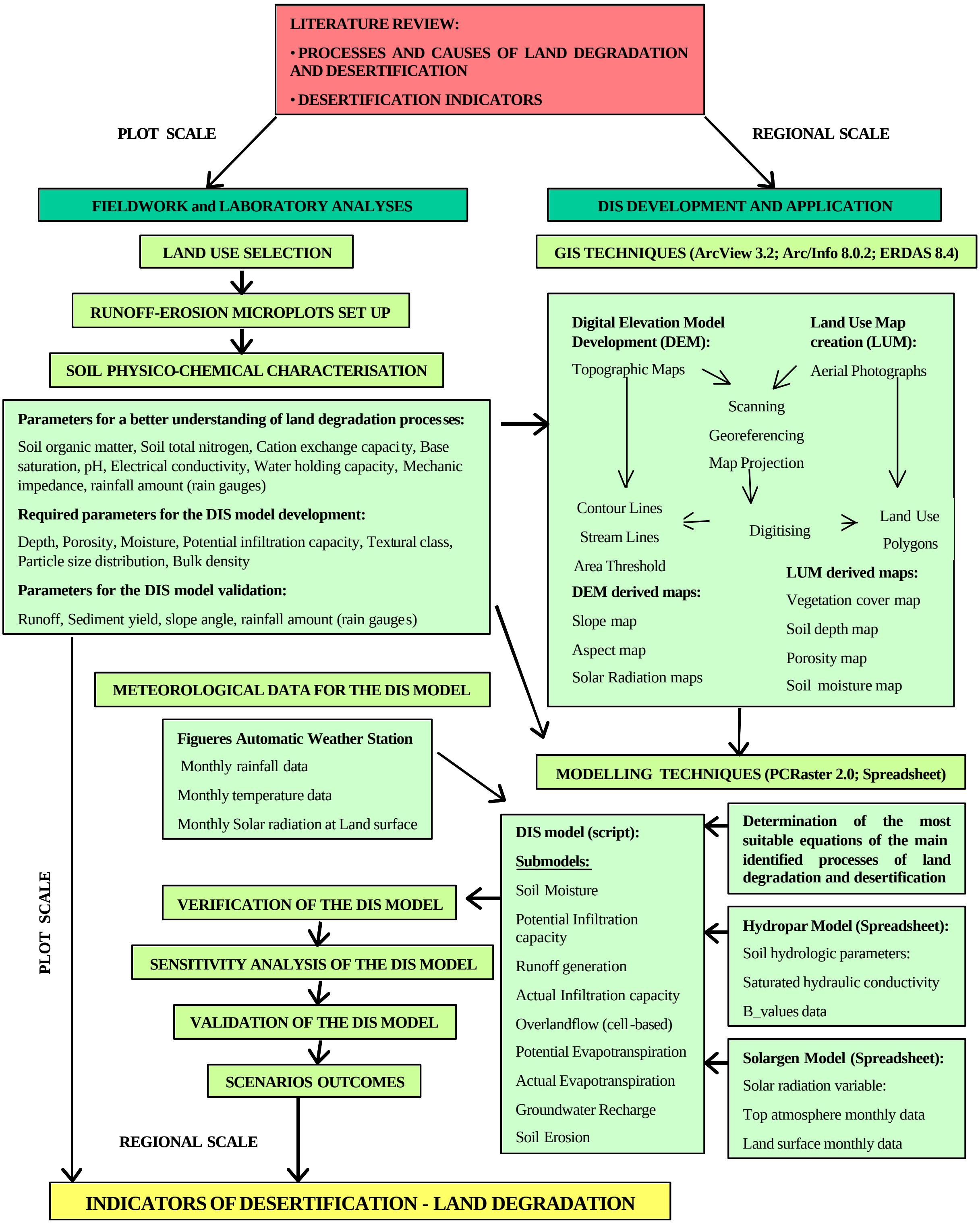


Figure 4.1. General flow diagram of the methodology

Regarding the second section, eight purposes are detailed: (a) the identification of the most relevant indicators of land degradation, (b) the development of a set of potential indicators of desertification in order to be applied in the development of the DIS, (c) the development of a spatial datasets for the study region, (d) the collection of relevant meteorological data, (e) the creation and verification of a process-based hydrological and soil-erosion model, (f) the sensitivity analysis of the verified model, (g) the validation of the model at the plot scale, and (h) the development of scenarios for land use policies of particular relevance in the region.

### 4.1.1. SCHEDULE OF THE OVERALL THESIS WORK

A brief summary of the different stages of the research work, conducted from summer 1999 to autumn 2003, is showed below:

| **Year** | **1** | **1** | **1** | **1** | **2** | **2** | **2** | **2** | **3** | **3** | **3** | **3** | **4** | **4** | **4** | **4** | **5** | **5** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **S** | ***su*** | ***au*** | ***wi*** | ***sp*** | ***su*** | ***au*** | ***wi*** | ***sp*** | ***su*** | ***au*** | ***wi*** | ***sp*** | ***su*** | ***au*** | ***wi*** | ***sp*** | ***su*** | ***au*** |
| *FW* | x | x | x | x | x | | | | | | | | | | | | | |
| *LA* | x | x | x | x | x | | | | | | | | | | | | | |
| *RW* | | | | x | x | | | | | | | | | | | | | |
| *LRB* | | | | | | x | x | | | | | | | | | | | |
| *DA* | | | | | | | x | x | x | | | | | | | | | |
| *LRL* | | | | | | | | | x | x | x | | | | | | | |
| *MD* | | | | | | | | | | | x | x | x | x | x | | | |
| *VE* | | | | | | | | | | | | | | x | x | | | |
| *SA* | | | | | | | | | | | | | | | x | x | | |
| *VA* | | | | | | | | | | | | | | | | x | x | x |
| *SO* | | | | | | | | | | | | | | | | | x | |
| *TW* | | | | | | | | | | x | x | | | | x | x | x | x |

S: season; su: summer; au: autumn; wi: winter; sp: spring; FW: field work; LA: laboratory analysis; RW: upgrade report writing; LRB: literature review and learning of new techniques; DA: data analysis; LRL: literature review (processes, causes, indicators of desertification); MD: model research and development; VE: verification of the model; SA: sensitivity analysis of the model; VA: validation fieldwork of the model; SO: runs of the scenario outcomes; TW: writing up of the thesis.

## 4.2. ASSESSMENT OF THE MOST IMPORTANT FACTORS OF LAND DEGRADATION AT PLOT SCALE

### 4.2.1. STUDY OF THE GLOBAL PROCESSES AND CAUSES OF LAND DEGRADATION AND THE SPECIFIC ONES FOR THE TARGET AREA

An extensive literature review of the main processes as well as human-induced and natural causes of land degradation and desertification from global to local scale, has been conducted in chapter two. This general review, will allow the identification of the main processes and causes of land degradation which are taking place in the Serra de Rodes catchment. Therefore, this identification will be the first step for the assessment of the main processes of land degradation in the target area.

### 4.2.2. RESEARCH STRATEGY

The assessment of the most important factors of land degradation at plot scale involved extensive experimental fieldwork and the associated laboratory analyses, which were conducted over one year, with seasonal observations, from summer 1999 to spring 2000.

#### 4.2.2.1. Selection of the representative environments of the study area

Aerial photograph interpretation, topographic maps and field-survey techniques were used to recognise eleven environments along two altitudinal transects in the study catchment. These environments were chosen in order to be representative of the different land uses in the study area, the Serra de Rodes catchment (see chapter one). Moreover, the easy accessibility to the site was also taken into account.

The selected environments are: a pine-tree plantation (*Pinus pinea, Pinus halepensis*), two dense cork tree and a sparse cork tree woodland (*Quercus suber*), two dense and one cleared shrublands (*Cistus monspeliensis, Cistus albidus, Lavandula stoechas, Calicotome spinosa, Ulex parviflorus, Erica arborea,...*), two olive groves (*Olea europaea*), one cultivated and one abandoned less than 5 years ago, a cultivated vineyard (*Vitis vinifera*) and a recently abandoned vineyard which will be considered as a bare soil environment, since all the vegetation was removed and the area was subsequently tilled.

Then, the eleven environments were grouped into nine land uses in order to assess the role of each different land use on the main processes of land degradation and desertification. Hence, the land uses taken into account in the analysis are: dense cork trees (DCT), sparse cork trees (SCT), dense shrubland (DS), sparse shrubland (SS), cultivated olive trees (COT), abandoned olive trees (AOT), cultivated vineyard (CV), reafforested pine trees (PTA), and bare soil (BS). Furthermore, for the development of the DIS model, the area covered by roads in the study site, was considered as another land use (ROAD), since it will definitively condition the overland flow as well as the flow drainage direction of the entire site.

### 4.2.3. ASSESSMENT OF THE ROLE OF LAND-USE/COVER ON THE MAIN SOIL PHYSICO-CHEMICAL PROPERTIES

The assessment of the effect of land-use cover on the main soil physico-chemical properties was carried out measuring: water holding capacity (WHC), pH, electrical conductivity (EC), bulk density (BD), soil moisture content (M), mechanical impedance (MI), exchangeable bases (V), soil organic matter (SOM) and total nitrogen (N), cation exchange capacity (CEC), particle size distribution (PSD) and textural class (TC), soil depth (SD), soil porosity (poros), and then comparing the results from the several selected environments. All analyses were made in duplicate by using conventional analytical methods (Ministerio de Agricultura, 1986).

In general soils of each environment were sampled at the beginning and end of every study season, in order to study the temporal variability of the relevant edaphic parameters. However, the TC and PSD, SD, poros as well as the CEC of the soils were determined only once, considering their limited variability throughout the one-year study period.

Soil samples were taken at 0-10 cm depth, at three different distances from each microplot for representative results. In the laboratory, samples were air dried (at 20ºC), mixed and sieved at 2 mm diameter. The organic residues and stones found in the soil sample were removed. As a result, a uniform soil sample was obtained per each environment.

#### 4.2.3.1. Soil organic matter content

The soil organic matter content (SOM) is defined as the sum of all natural and thermally altered biologically derived organic material found in the soil or on the soil surface irrespective of its source, whether it is living or dead, its stage of decomposition, but excluding the aboveground portion of living plants (Baldock and Nelson, 2000). This parameter is an important constituent of the soil. It contributes to the nutrient retention capability of the soil through cation exchange capacity (CEC) and the ability to complex or chelate metals. It also has an important effect on soil physical properties. Organic matter serves as a binding agent to help aggregate soils. Thus organic matter influences soil structure and porosity. Structure and porosity in turn influence infiltration and percolation of water through the soil and the retention or storage of water in soils. In addition, it also serves as a source of nutrients to plants and microorganisms, and it is considered an important part of the global carbon cycle. Therefore, SOM affects the soil biological, chemical and physical properties in many different ways.

There are a number of procedures that are used to measure soil organic matter. Generally they can be categorized as either wet or dry combustion techniques, and both techniques involve the oxidation of the organic carbon and the quantification, directly or indirectly, of the $CO_2$ produced. The analytical method used for determining the soil organic matter content is the Wet Oxidation Method or commonly called Walkley-Black Method (Walkley-Black, 1934), considered the standard method for estimating organic matter in soils, which uses chemical oxidizing agents to oxidize the organic carbon (Nelson and Sommers, 1982). This procedure measures the easily oxidizable organic carbon (OC) by oxidation with potassium dichromate ($K_2Cr_2O_7$).

#### 4.2.3.2. Soil total nitrogen

The determination of the soil total nitrogen was based on the Kjeldahl Total Nitrogen (TKN) method (Kjeldahl, 1883), which is based on the wet oxidation of soil organic matter and botanical materials using sulphuric acid and a digestion catalyst and conversion of organic nitrogen to the ammonium form (Bremner and Mulvaney, 1982). Ammonium is determined using the diffusion-conductivity technique. The procedure does not quantitatively digest nitrogen from heterocyclic compounds (bound in a carbon ring), oxidized forms such as nitrate and nitrite, or ammonium from within mineral lattice structures. The method has a detection limit of approximately 0.001% N and is generally reproducible within 8%.

#### 4.2.3.3. Cation-exchange capacity

The cation-exchange capacity (CEC) of a soil is simply a measure of the quantity of sites on soil surfaces that can retain positively charged ions (cations) by electrostatic forces (Sumner and Miller, 1996). Cations retained electrostatically are easily exchangeable with other cations in the soil solution and are thus readily available for plant uptake. Thus, CEC is important for maintaining adequate quantities of plant available calcium ($Ca^{2+}$), magnesium ($Mg^{2+}$), sodium ($Na^+$) and potassium ($K^+$) in soils. While a soil with a higher CEC may not necessarily be more fertile, when combined with other measures of soil fertility, CEC is a good indicator of soil quality and productivity. Soil CEC is normally expressed in units of charge per weight of soil. Two different, but numerically equivalent sets if units are used: meq/100g (milliequivalents of charge per 100 g of dry soil) or cmol/kg (centimols of charge per kilogram of dry soil). The method used for this analysis is the determination of CEC at pH 7 with Ammonium Acetate (Chapman, 1965).

#### 4.2.3.4. Base-saturation percentage

Base-saturation percentage (V) is the ratio of basic cations (calcium, magnesium, potassium and sodium) expressed as a percentage of the CEC. It indicates the proportion of exchangeable cation sites that are filled by the exchangeable basic cations (Schollenberger and Simon, 1945). Base Saturation is the extent to which the adsorption complex of a soil is saturated with exchangeable cations other than hydrogen or aluminium. It is expressed as a percentage of the total CEC. Soils with low values of base saturation are considered to be leached and are often acid, whereas neutral and alkaline soils tend to have high base saturation. This analysis is based on the same methodology used for the CEC determination, although it only consists of the extraction with ammonium acetate.

The exchangeable cations ($Ca^{2+}$, $Mg^{2+}$, $K^{+}$, $Na^{+}$) are determined by atomic absorption spectrophotometry (AAS), an analytical technique used to measure a wide range of elements (Figure 4. 3.). Measurements are made separately for each element of interest in turn to achieve a complete analysis of an object, and thus the technique is relatively slow to use. However, it is very sensitive and it can measure trace elements down to the parts per million level, as well as being able to measure elements present in minor and major amounts.

### 4.2.3.5. pH or soil reaction

Soil reaction is a numerical expression of the relative acidity or alkalinity of a soil. The pH indicates the hydrogen ion (H+) concentration of a solution, a measure of its acidity. The pH is defined as the negative logarithm of the concentration of $H^+$ ions. Because $H^+$ ions associate with the water molecules to form hydronium ($H_3O^+$) ions, pH also is often expressed in terms of the concentration of $H_3O^+$. The determination of $H^+$ also provides a measurement of $OH^-$ in solution. Acid solutions have a pH ranging from 0 to 6.9, and basic solutions can have a pH ranging from 14 down to, but not including 7.1. Acids with lower numbers and bases with higher numbers are stronger. Solutions with a pH equal to 7 are considered neutral.

Soil pH is probably the single most important chemical characteristic of a soil. It provides information about associated soil characteristics, such as available plant nutrients. Strongly acid or more acid soils have low extractable calcium and magnesium, and a high solubility of aluminium, iron and boron. In addition, these soils have a possibility of organic toxins and generally have a low availability of nitrogen and phosphorus. At the other extreme are alkaline soils. Calcium, magnesium, and molybdenum are abundant with little or no toxic aluminium, and nitrogen will be readily available. If the pH is above 7.9, the soils may have an inadequate availability of iron, manganese, copper, zinc, and especially of phosphorus and boron. Soil reaction is one of several properties used as a general indicator of soil corrosivity or its susceptibility to dispersion. Soils that have pH $< 5.5$ are likely to be corrosive to concrete. Soils that have pH $> 8.5$ are likely to be highly dispersible, and piping may be a problem.

The measuring methods for pH values include using an indicator reagent, the metal electrode methods and the glass electrode method. The glass electrode method is considered to be the standard measuring method (Soil Survey Laboratory Staff, 1996). This method, which is based on the use of a pH-meter, is the one conducted for the pH determinations of the soils of the study area. The pH-meter consists of a glass electrode and a reference electrode (Figure 4.4.). It allows the pH value of the sample to be obtained by measuring the potential difference between the two electrodes with a potential difference meter.

#### 4.2.3.6. Electrical conductivity

The electrical conductivity (EC) of a saturation extract is the standard measure of salinity (Salinity Laboratory Staff, 1954). EC is related to the amount of salts soluble in the soil and is dependent on the activity and type of dissolved ions and the solution temperature. The higher the concentration of salt in a solution, the higher will be the electrical conductance (the reciprocal of resistance). The standard international units of measure are decisiemens per metre (dS/m) corrected to a temperature of 25ºC. The measurement of the soil EC is conducted by the electrical conductivitymeter (Rhoades, 1982).

#### 4.2.3.7. Water-holding capacity

Water-holding capacity (WHC) is dependent upon the soil organic matter content and soil particle size. Normal field soil water-holding capacity is 60-80% of its total capacity; that is, 60-80% of the water filled pore spaces are filled. This corresponds to the optimal biological activity for water-holding capacity. When the water-holding capacity falls below 55-60%, organisms could suffer from dryness; and when the capacity is over 80%, they begin to suffer from a depletion of soil oxygen. How much water a soil can hold is very important for plant growth. Soils that can hold a lot of water support more plant growth and are less susceptible to leaching losses of nutrients and pesticides. All of the water held by soil is not necessarily available for plant growth. The method described by Klute (1986) was used here.

### 4.2.3.8. Mechanical impedance or penetrability

Penetration resistance is the capacity of the soil in its confined state to resist penetration by a rigid object. Penetration resistance depends strongly on soil moisture. A standard instrument is the pocket penetrometer shown in Bradford (1986), which measures gauging the structure of soil by measuring the degree of soil compaction at depths (Figure 4.5.). Penetration is expressed in units of pressure. Therefore a manual static penetrometer ranging from 0 to 6 kg/cm$^2$ and a flat-end tip of 0.5 cm is used for the determination of the soil mechanical impedance. The procedure is based on applying a constant force with the penetrometer to a depth of 1 cm in the soil. In order to obtain good results, this process is conducted 5 times addressing the North, South, East, West and centre of the interior area of each microplot. The penetrability of the soil related to each environment is the average of these five values.

### 4.2.3.9. Soil textural class and particle size distribution

Soil texture is considered to be one of the least changeable features of a soil, since it controls many soil physical, chemical, and biological characteristics. Differences in texture between two soils can produce differences in drainage rates, the amount of water available to plants, aeration, decomposition rates, soil organic matter, temperature, susceptibility to wind and water erosion, and the rate at which pollutants in groundwater are transported. Therefore, soil texture plays a major role in determining the type of plant and animal community that is found above and below ground, and its determination is essential in any soil related study.

Soil texture refers to the weight proportion (relative proportion by weight percentage of sand, silt and clay) of the mineral soil separates for particles less than two millimetres (mm) as determined from a laboratory particle-size distribution (Soil Survey Division Staff, 1993; Ministerio de Agricultura, 1986). Natural soils are comprised of soil particles that vary in size. Therefore, particle-size groups or soil separates are categorized as: sand (the coarsest); silt; and clays (the smallest/finest). The present analysis considers the classification for the proportions of sand, silt and clay particles defined by the International Society of Soil Science (ISSS):

| | |
|---|---|
| Sand: | **2 mm < ø < 0.2 mm** |
| Fine sand: | **0.2 mm < ø <0.02 mm** |
| Silt: | **0.02 mm < ø < 0.002 mm** |
| Clay: | **ø < 0.002 mm** |

Particle size analysis was carried out with 22 soils samples collected from three different points (for representative results) around the 22 runoff-erosion microplots. The sand, silt and clay fractions were determined using the Kilmer and Alexander (1949) pipet method, which is considered by the Soil Conservation Service (Soil Survey Laboratory Staff, 1996) as the standard method, since it is reproducible in a wide range of soils.

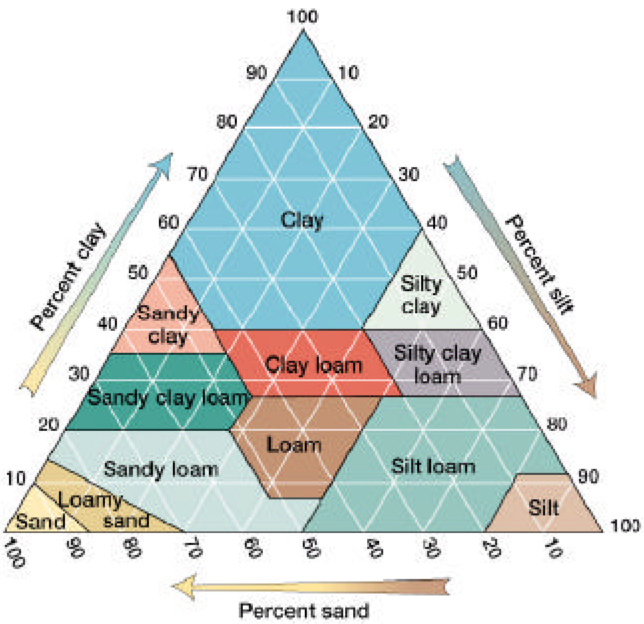


Figure 4.2. USDA Textural triangle (Dingman, 2002a)

There exist multiple possible combinations once the percentages of clay, silt and sand of a soil are available, since they can be grouped in a few classes of particle sizes or textural classes. These specific issues provide an idea of the synthesis and facilitate the utility of the information. The United States Department of Agriculture (USDA) textural triangle is a graphical representation of the 12 soil textural classes (Figure 4.3.).

#### 4.2.3.10. Soil depth

The soil depth is defined as the vertical distance between the surface and the parent material of a soil, which is impermeable to roots or percolating water (Soil Survey Division Staff, 1993). Insufficient soil depth for adequate root development is a physical property of the soil that can affect plant growth and crop yields. Soil properties, which prevent roots from penetrating the subsoil can be among others: hard pans, platy structure and stratified layers. Deep soils can hold more plant nutrients and water than can shallow soils with similar textures. Depth of soil and its capacity for nutrients and water frequently determine the yield from a crop, particularly annual crops that are grown with little or no irrigation. Plants growing on shallow soils also have less mechanical support than those growing in deep soils.

Depth is measured from the soil surface, which is the top of the mineral soil, or for soils with an O horizon, the soil surface is the top of the part of the O horizon that is at least partially decomposed (Soil Survey Staff, 1993). Once the soil material is removed from the topsoil layer downward to the bedrock by a spade and/or auger, soil depth is directly determined for each environment using a measuring tape.

#### 4.2.3.11. Bulk density

According to Porta et al. (1994a), the bulk density of soil is defined as the relationship between the mass and volume of an unmodified sample of dry soil, meaning that the sample has been taken in the field without altering any of its edaphic properties. This parameter is an indicator of the level of porosity or compaction and of the structural properties of a soil, which strictly control the hydrological properties of the soil profile. Bulk density is highly dependent on soil conditions at the time of sampling. Changes in soil swelling due to changes in water content will alter density. Soil mass remains fixed, but the volume of soil may change as water content changes (Soil Survey Laboratory Staff, 1996). The methodology for bulk density determination used here was the described by Blake and Hartge (1986). A single sample was taken from each of the 22 microplots both at the beginning and end of each season of the year. Samples were collected in the following way:

1. Hammer a calibrated cylinder of 5 cm longitude and 5 cm diameter into the soil of the external area of each microplot.
2. Remove the soil around the cylinder with a small shovel and carefully place it above the shovel, levelling off the excess of soil from the superior part and covering it with a plug. The same procedure will be done for the inferior part of the cylinder, maintaining the sample in the same condition as is was in the soil.

#### 4.2.3.12. Soil moisture content

The moisture content of the soil will be one of the main sub-models of the DIS and since it controls, and is controlled by, many aspects of the DIS, it is an important variable for measurement.

Soil water is a highly dynamic entity, exhibiting substantial variation in both time and space (Or and Wraith, 2000). Changes in soil-water content and its energy status affect many soil mechanical properties including strength, compactibility and penetrability. Water in the soil occupies pore spaces that arise from the physical arrangement of the particulate solid phase, competitively and often concurrently with the soil gas phase. The liquid phase characteristics affect the soil gaseous phase and the rates of exchange between these phases, as well as other important soil properties such as the hydraulic conductivity.

The determination of soil moisture achieved here was by gravimetric assessment. The procedure for this determination is exactly the same as the one used for the bulk density. Soil moisture content is calculated as the difference in weight between the soil in its natural conditions and its weight after drying it in the oven at a temperature of 105ºC over 24 hours. The soil moisture content is determined as:

$$M = [(Pf - T) - (Ps - T)] / (Ps - T) \tag{4-1}$$

M = actual soil moisture content (g water/g soil)
Pf = fresh weight (g)
Ps = dry weight (g)
T = cylinder tare (g)

*Pf* refers to the total weight of the soil sample, including the porespace filled with water. On the contrary, *Ps* only refers to the weight of the soil, excluding the porespace, since it is filled with air. The difference between *Pf* and *Ps* accounts for the weight of the porespace filled with water. Therefore, the moisture is expressed as grams of water per gram of soil. However, in this study soil moisture will be expressed in $m^3$ water/ $m^3$ soil.

#### 4.2.3.13. Soil porosity

The total porosity or total pore space of a soil is simply the amount of empty space located between soil particles. This parameter is of great importance since it normally represents 50% of the total volume of a soil (45% of mineral matter and 5% of organic matter on average).

Soils with high percentages of sand have less total porosity than soils with more clay particles. Although sandy soils are less porous, they tend to possess a greater amount of large sized pores, called macropores. Macropores provide soil aeration and efficiently drain excess water, and are often caused by the increase of root growth and animal burrows. On the other hand, there are disadvantages of a dominance of large grained soil particles in a soil, since extremely sandy soils, typically hold much less water than other soil types. Like bulk density, porosity is constant over the time periods considered in most hydrological analyses. However, in many soils, it decreases with depth due to compaction and the development of macropores by biologic activity near the surface (Dingman, 2002a). For this one-year study, this parameter is considered constant and has been measured only once.

With reference to the calculation of the soil total porosity, this parameter is inversely related to bulk density through the expression:

$$P = 1 - (BD / D_s) \tag{4-2}$$

P = total soil porosity (dimensionless: $m^3$ porespace/$m^3$ soil)
BD = bulk density of the soil ($g/cm^3$)
$D_s$ = particle density (usually considered as 2.65 $g/cm^3$, although it depends on the mineralogy of the soil)

Hence, the total soil porosity is usually determined by measuring BD and assuming an appropriate value for the $D_s$.

#### 4.2.4. ASSESSMENT OF LAND-DEGRADATION PROCESSES AT PLOT SCALE

The assessment of the main land-degradation processes will be based on the monitoring of runoff-erosion microplots installed in different environments representative of the several land uses in the Serra de Rodes catchment. The recording of rainfall data after each rainfall event as well as the determination of the vegetation cover of each environment, will be essential for evaluating their effect on the runoff and soil erosion processes in the target area.

#### 4.2.4.5. Runoff and soil erosion

With the aim of evaluating the role of land use/land cover on land-degradation processes such as runoff generation and sediment yield, replicated experimental runoff-erosion microplots, less than 1 $m^2$ (Ruíz-Flaño, 1993; Bochet et al., 1995), were installed in each of the selected environments. A total of 22 experimental microplots differing in slope, orientation, stoniness and vegetation cover percentage, were monitored throughout one year of fieldwork.


Figure 4.3. Runoff deposit (left) and sediment yield container (right) of a runoff-erosion microplot after a rainfall event.

Each microplot was bounded with strips of zinc sheeting and trenched around to lead runoff water and eroded soil away from upslope to downslope. Sediment yield was collected in a container at the soil surface and runoff water was drained into a receptacle installed into the ground. Both containers were cleared of sediment and water on a rainfall-event basis (Figure 4.3), in order to measure and evaluate their rates.

The bounded design of the microplots accounts for a closed system where sediment transported downslope is not replaced by upslope material, with the result that the more readily erodible material tends to become depleted with time (Vacca et al., 2000; Romero-Díaz et al., 1999). However, these microplots are generally thought to work well for one year with a limited number of events, which was the case for this study. The main advantage of the selection of these microplots was their low cost and easy installation in the field.

#### 4.2.4.6. Vegetation cover

The vegetation cover of the soil will be considered an essential factor in the development of the DIS model, since it plays an important role on many soil-water processes. The vegetation cover of the several land uses in the Serra de Rodes catchment was determined twice during the study period, in summer 1999 and spring 2000. Two determinations were conducted in order to know the difference in vegetation cover between wet and dry seasons.

Measurements were specifically focussed on the vegetative cover of the area bounded by each microplot. These estimations were only based on visual determinations without quantitative measurements. Both live and dead vegetation were considered together as the total vegetation cover protecting the soil surface. They were visually evaluated as the percentage of ground surface cover. Regarding the aerial vegetation and accounting for its role on the interception of the rainfall and for instance its protection to the soil, the determination of the percentage of this parameter was measured on the basis of the area of each microplot covered by the woody canopy. However, most of the microplots were not affected by aerial vegetation cover (Figure 4.4). Transect techniques were not used because the areas of each microplot (1 $m^2$) were considered too small for this kind of determination.

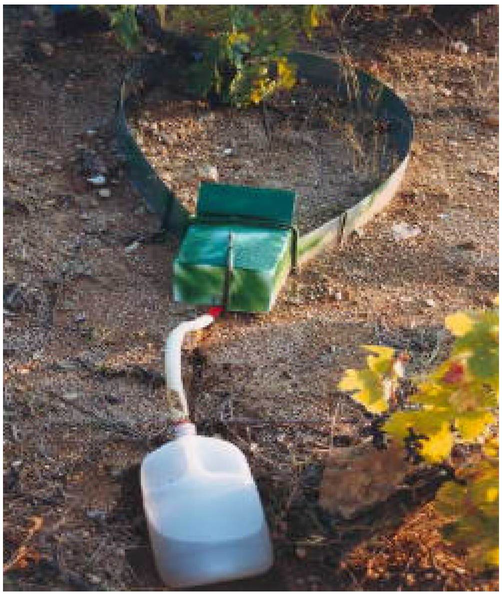


Figure 4.4. Example of a runoff-erosion microplot set up in a vineyard (left) and shrub environment (right).

#### 4.2.4.7. Soil infiltration capacity

The potential infiltration capacity is considered one of the main components of the DIS model and field data on this parameter will be used as input of this sub-model. Infiltration is the process by which water arriving at the soil surface enters the soil. The values are usually sensitive to near-surface conditions as well as to antecedent water state, and therefore they are subject to significant change with soil use and management as well as time. Infiltration capacity of a soil is defined as the maximum rate at which infiltration can occur, and its value changes during the infiltration event (Soil Survey Division Staff, 1993). This parameter can be directly measured in the field over a small area, using a *ring infiltrometer* (Dingman, 2002a). The area is defined by an impermeable boundary, usually a cylindrical ring extending several centimetres above the surface, and sealed at the surface or extending several centimetres into the soil.

Figure 4.5. Example of a double-ring infiltrometer used for field determinations

The infiltration rate is the rate at which water enters the soil from the surface and can be obtained by (a) measuring the rate at which the level of ponded water decreases, (b) measuring the rate at which water has to be added to maintain a constant level of ponding, or (c) solving a water-balance equation for the ponded surface:

I (t) = [W – Q – (? H * A)] / ?t (4-3)

I (t) = Infiltration rate average over a period of measurement, ?t ($m^3$/h)

W = Volume of water applied during ?t ($m^3$)

Q = Volume of ponded water removed from the plot during ?t ($m^3$)

?H = Change in ponded-water level during ?t (m)

A = Area covered by the infiltrometer ($m^2$)

?t = Period of measurement (h)

Infiltration rates during the early stages of the process are usually high and gradually decrease to a nearly constant value; it is this constant value that is usually taken as the saturated hydraulic conductivity of the near-surface soil.

However, because water infiltrating into an unsaturated soil is influenced by both capillary (pressure) and gravity forces, the water applied within an infiltrometer ring moves laterally as well as vertically. Thus, the measured infiltration rate exceeds the rate that would occur if the entire surface were ponded. Therefore, in order to reduce this effect, the *double-ring infiltrometer* was used in the field for all the measurements concerning the environments related to the several land uses in the study area (Figure 4.5.). With this device, the area between the two rings acts as a “buffer zone”, and measurements only on the inner ring are used to calculate the infiltration rate.

**4.2.4.8. Rain gauges**

Several rain gauges of 5 and 25 litre volume were randomly set up in the Serra de Rodes catchment, in order to cover the overall rainfall range characteristic in the study area. Rainfall amount was quantified after each rainfall event. Rain gauges of 5 litre volume consisted of a plastic container with a funnel of 30 cm diameter, standing on top of a 1 metre long wooden support (Figure 4.6). Rain gauges of 25 litre volume which consisted of a container of 70 cm and a 30 cm diameter funnel, were directly placed on the ground (Figure 4.6).

In addition to the rain gauges data, monthly rainfall amount and intensity from 1999 and 2000 was provided by the Figueres automatic weather station (AWS), located nearby the target area.

Figure 4.6. Example of a rain gauge of five-litre volume set up in a shrub environment (left); and one of 25 litre placed at the uppermost part of the study catchment (right).

### 4.2.5. DEVELOPMENT OF SOIL QUALITY INDICES

The development of indices of soil quality will be conducted on the basis of a principal component factor analysis (PCA) in order to reduce the number of variables and also to classify the variables. Thus, the PCA will allow the simplification and summary of the contribution of the overall set of measured soil physico-chemical variables as well as soil erosion, overland flow and vegetation cover, by grouping them into several factors or indices. The procedure of the PCA is detailed in the paper Paniagua et al. (1999).

## 4.3. DEVELOPMENT AND APPLICATION OF THE DIS MODEL

The development of a desertification indicator system (DIS) for a semi arid Mediterranean catchment, aims at the formalization of desertification indicators as process-based models and the identification of present and future threats to effectively programme measures to combat desertification. The model will be on a monthly time scale due to the availability of the meteorological data in the study area, and it will be run over a period of one year (see chapter five for more detail).

### 4.3.1. IDENTIFICATION OF THE MOST RELEVANT INDICATORS OF LAND DEGRADATION

The identification, description and study of indicators of desertification and land degradation, from local to global scale, was conducted through an extensive literature review in chapter three.

### 4.3.2. DEVELOPMENT OF A SET OF POTENTIAL INDICATORS OF DESERTIFICATION IN ORDER TO BE APPLIED IN THE CREATION OF THE DIS

Once the assessment of the most important factors of land degradation at a plot scale has been conduced, these factors will be evaluated in order to determine whether they have the characteristics required for being good indicators and therefore could be considered for the development of the DIS. Moreover, from the overall set of identified relevant indicators of land degradation (chapter three), an assessment will be conducted in order to determine whether some of them are important for the study area and could be used in the creation of the DIS.

### 4.3.3. DEVELOPMENT OF SPATIAL DATASETS

The development of spatial datasets for the study region will be conducted on the basis of existing cartographic data and from which some indicators and model parameters could be derived in order to develop spatial indicators of desertification processes.

#### 4.3.3.1. Geographical information systems (GIS) techniques

Geographical Information Systems (GIS) are a powerful tool for managing resources, understanding and predicting complex and changing systems. A GIS comprises a collection of integrated computer hardware and software, which together are used for inputting, storing, manipulating, analysing and presenting a variety of geographical data (Peuquet et al., 1993).

The scanning of the topographic maps and aerial photographs, both at a scale of 1:5000, was conducted using an A3 scanner at 1200 dpi. The images were georeferenced using ERDAS 8.4. (ERDAS, 1997). The ERDAS software deals with geographical information system analysis (GIS) and remote sensing, and was selected for being most user-friendly and useful for this particular task in terms of management.

##### 4.3.3.1.1. Digital elevation model (DEM)

A digital elevation model (DEM) is a digital file (cartographic/geographic dataset) consisting of terrain elevations (in xyz coordinates) for ground positions at regularly spaced horizontal intervals (USGS, 2003). DEMs are derived from hypsographic data (contour lines) and/or photogrammetric methods using topographic quadrangle maps. Topographic maps, which are visual ways of showing information of the earth surface such as roads, rivers, elevation, etc., present the information only on a flat surface in two dimensions. Height (elevation) can only be shown in maps in the form of contour lines, which connect points of equal heights.

The DEM of the study catchment was developed using both the ArcView 3.2 (with Spatial Analyst 2.0) and Arc/Info 8.0.2 (TOPOGRID), GIS software packages. The DEM was created using the TOPOGRID command (ESRI 2000), which is an interpolation method specifically designed for the development of hydrologically correct DEMs from comparatively small, but well-selected elevation and stream data.

This ARC command generates an hydrologically correct grid of elevation from point, line and polygon coverages. It is based upon the ANUDEM program (Hutchinson, 1988, 1989). The first step in the creation of a DEM is the digitising of four features from the topographic maps:

1. Contour lines: contour map representing lines with the same elevation value (line shapefile: vector data). Every contour line has assigned its correspondent elevation value in the “Attribute Table”.
2. Stream lines: stream map of the several rivers located in the study area (line shapefile: vector data).
3. Area threshold: map of the boundary of the study catchment (polygon shapefile: vector data).
4. Points of the rivers intersecting with the edge of the area threshold (point shapefile: vector data).

Once the DEM is created, some maps can be directly derived from it. The slope map of the study area was needed in the DIS model when calculating the soil erosion output. In this context, the slope map was computed using the GRID command, which is the raster processing component of Arc/Info. The slope map could be derived in degrees or percentages from the DEM. However, for this particular project and according to the wash erosion model (Thornes, 1990), the slope map was created in degrees. In the same way the aspect map is also calculated from the DEM. The aspect map was needed for calculating the solar radiation maps, required for the potential evapotranspiration sub-model in the DIS template.

#### 4.3.3.1.2. Land-use map

The land-use map (LUM) was created on the basis of the scanned aerial photographs of the study area. For the LUM development, it was necessary to identify manually from the aerial photographs, the several land uses present in the catchment (10 land uses), and then digitise (ArcView 3.2) them in the study area as polygons, assigning to each one, its identification value (number from one to ten) in the attribute table.

The LUM was the basis for the creation of a set of maps also required in the development of the DIS model, such as: vegetation cover, potential infiltration, soil moisture, soil depth and soil porosity map. These maps were created using the MAP CALCULATOR and MAP QUERY in ArcView 3.2.

#### 4.3.4. METEOROLOGICAL DATA REQUIRED FOR THE DIS MODEL

Meteorological data was obtained from an Automatic Weather Station (AWS), located in the vicinity of the study area at Figueres (42º 15’ 59’’N, 2º 57’ 35’’E). Precipitation was the most important meteorological parameter, since the model aims to assess its role on runoff generation and sediment yield (model outputs) in the ten selected land uses in the catchment. Thus, three different rainfall scenarios from different years were tested in the model: dry (1980), standard or normal (1997) and wet (1993) year, with amounts ranging from 430.9 and 585.4 to 995.9 mm respectively. Additionally, several manual rain gauges were randomly located in the study area, in order to obtain data from specific rainfall events and relate them to the outputs of the runoff-erosion microplots. Temperature and solar radiation data were needed in the potential evapotranspiration submodel. However, solar radiation measures, which consisted of a single location determination (AWS), were used for the creation of maps of solar radiation at the land surface for the entire study catchment (see chapter five).

#### 4.3.5. CREATION OF A PROCESS-BASED HYDROLOGICAL AND SOIL-EROSION MODEL

The incorporation of a modelling component to a GIS was needed for the prediction of potential outcomes and evaluating alternatives. Thus, the development of the DIS not only required the use of GIS packages such as ArcView 3.2. and Arc/Info 8.0., but also spatial modelling tools such as spreadsheet modelling and PCRaster 2.0 software, which allowed dynamic process modelling over space and time through a generic scripting modelling language (Van Deursen and Wesseling, 1997).

#### 4.3.5.1. Determination of the most suitable mathematical equations of the identified processes of land degradation and desertification

The DIS model is composed of several interrelated submodels, which describe the main land-degradation processes identified in the study area. These components of the model are: soil moisture content, potential infiltration capacity, runoff generation, actual infiltration capacity, overlandflow, potential evapotranspiration, actual evapotranspiration, groundwater recharge and soil erosion. The DIS model was created integrating the mathematical equations of these processes in the PCRaster 2.0 software, which combines GIS tools and spatial modelling. The identification of the most suitable expression representing each one of these processes (indicators) was conducted through literature review. These expressions not only had to be as simple as possible but they had also to be well established and reliable, with regard to the relevance of the process, and the spatial and temporal scale of the project. This process is detailed in chapter five.

#### 4.3.5.2. Verification of the DIS model

The verification process of the DIS was conducted through running the model in PCRaster 2.0 software, once completed. This process aimed to find any conceptual and/ or technical errors in the model. Conceptual errors were clear when looking at the outputs results, whereas the technical ones were directly identified when running the model. The first kind of errors mean that conceptually, a certain equation or equations selected for composing the several submodels of the DIS template is/are not correct, needing revision or change in order to accomplish the objectives set up for the project. On the other hand, the technical errors are normally related to specific mistakes in the script, and although sometimes the software can easily identify them, for some others it is a very difficult and time-consuming task. Additionally, the verification of the model was also performed through the sensitivity analysis of the DIS and the matching of parameter sensitivity and model behaviour against the modeller's understanding of the system being modelled.

### 4.3.6. SENSITIVITY ANALYSIS OF THE DIS MODEL

The sensitivity analysis of the DIS model was conducted in order to assess the role of rainfall in land-degradation processes such as runoff generation and soil erosion. The analysis consisted of running the DIS model under three different rainfall scenarios (wet, normal and dry). The assessment and comparison of the outputs allowed the determination of the effect of rainfall on these soil-water processes. At the same time, several parameters composing the equations of the sub-models were tested for their relevance on the sensitivity of the main processes of land degradation. Therefore, the analysis not only allowed the determination of the most important parameters in the model, but it also contributed on the assessment of the good performance of the DIS model (verification).

Thus, the sensitivity analysis was an essential factor in the DIS model and a whole chapter is dedicated to describe the objectives, procedure, results and conclusions of this particular assessment (see chapter seven).

### 4.3.7. VALIDATION OF THE DIS MODEL

In essence, the validation of the DIS model allowed the assessment of the reliability of the model outcomes. The run of the DIS was individually performed on a one-cell basis (5 $m^2$) per each land use in the study site, using field monthly soil loss, runoff and rainfall data recorded during the study period. This analysis allowed the comparison of the outcomes of the DIS model, associated with the main processes of desertification and the experimental data available from field determinations, and therefore the assessment of the consistence and coherence of the DIS. The exact procedure conducted for this analysis is detailed in chapter eight, where the results are presented and discussed.

### 4.3.8. SCENARIO OUTCOMES OF THE DIS MODEL

This analysis consisted of the development of several scenarios in order to enable the prediction of potential desertification risk and the assessment of alternatives for management, combining indicators of both the processes of degradation and its impacts. These scenarios will represent the application of particular policies, bearing in mind the ones of particular interest in the region such as: land abandonment, methods for fire prevention and control through the management of vegetation characteristics. In general the procedure consists of:

1. Setting particular land use scenarios for the whole catchment, specifying the percentage of vegetation coverage (e.g. 70% of the area covered by cultivated field and 30% by shrubs).
2. Run of the model under the assigned conditions.
3. Analysis and comparison of the different scenario outcomes with regards to simulated runoff and erosion in order to gain a better insight into the land degradation and desertification processes in the study site, as well as to describe possible political actions in terms of sustainable management of the natural resources.

Further explanation, the results and discussion of this assessment is detailed in chapter eight.

# 5. RESULTS OF PLOT SCALE ANALYSES, IDENTIFICATION OF LAND DEGRADATION INDICATORS AND DEVELOPMENT OF SPATIAL DATASETS



## 5.1. INTRODUCTION

This chapter is structured in relation to the main specific objectives detailed in chapter one. On the one hand, it seeks to present the results of the several analyses implicated in the assessment of the most important factors of land degradation at plot scale in the target area, and on the other hand, the tasks involved in the development and application of the DIS model. The former specific objective will entail the assessment of the role of land use-cover on the main soil physico-chemical properties, the evaluation of land degradation processes at plot scale and the development of soil quality indices. The latter specific objective will involve the creation of a set of potential indicators of land degradation, and the development of spatial datasets, whereas the rest of phases related to this objective, such as the creation of a process-based hydrological and soil-erosion model, the collection of relevant meteorological data, as well as the model sensitivity analysis, validation and creation of the scenario outcomes, will be explained in more detail in the forthcoming chapters.

## 5.2. ASSESSMENT OF THE MOST IMPORTANT FACTORS OF LAND DEGRADATION AT PLOT SCALE

### 5.2.1. IDENTIFICATION OF THE MAIN PROCESSES OF LAND DEGRADATION IN THE TARGET AREA

The identification of the main processes of land degradation potentially related to the target area, the Serra de Rodes catchment, has been carried out on the basis of the reviewed processes and causes of land degradation and desertification (see chapter two). As a result, although soil degradation is considered the main process of land degradation in the study area, the following processes will be directly or indirectly assessed: (a) the degradation of the vegetative cover, (b) physical soil degradation in terms of soil compaction and crusting, (c) chemical soil degradation and specifically soil acidification, (d) biological soil degradation in terms of loss of soil organic matter and (e) soil water erosion processes.

### 5.2.2. ASSESSMENT OF THE ROLE OF LAND-USE/COVER ON THE MAIN SOIL PHYSICO-CHEMICAL PROPERTIES

Results from soil characterisation are shown in Table 5.1., where the value of each parameter is the mean of four field and/or analytical determinations per environment.

Unexpectedly, pH values seem neither to be dependent on soil organic matter content (SOM), nor on the land-use sequence. Instead, pH results from both studied seasons, revealed a similar pattern among all the environments. Therefore, pH values are likely to be related to the soil parent material, characterized by acid rocks in the whole study area (Dunjó et al., 2003). In general pH is slightly higher in winter in all environments. To some extent this could be related to the season, since winter is characterized by wetter conditions than in summer, and therefore the solution of the soil could produce a dilution effect resulting in more alkaline pH values (Dunjó et al., 2003).

Table 5.1. Means (N= 4) of physical and chemical soil parameters in the different land use types, comparing both seasonal study periods (s: summer and w: winter)

| **Soil parameter** | | **Environment current land use type** | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **A** | | | **B** | | **C** | | | **D** | | |
| | **S** | **DCT** | **DCT** | **PTA** | **AOT** | **SCT** | **DS** | **SS** | **DS** | **COT** | **BS** | **CV** |
| **Chemical** | | | | | | | | | | | | |
| pH ($H_2O$, 1:2.5) | s | 6.36 | 5.7 | 6.47 | 6.25 | 6.11 | 6.57 | 6.15 | 6.11 | 6.75 | 6.4 | 5.17 |
| | w | 6.21 | 6.2 | 6.67 | 6.67 | 6.13 | 6.68 | 6.2 | 6.31 | 6.78 | 6.46 | 5.23 |
| EC (dS/m) (1:5) | s | 0.24 | 0.17 | 0.35 | 0.08 | 0.08 | 0.08 | 0.09 | 0.12 | 0.13 | 0.07 | 0.08 |
| | w | 0.10 | 0.15 | 0.13 | 0.09 | 0.07 | 0.08 | 0.06 | 0.07 | 0.07 | 0.09 | 0.07 |
| M (%) | s | 15.27 | 12.67 | 8.30 | 8.12 | 7.95 | 7.06 | 7.97 | 12.93 | 6.92 | 7.69 | 7.31 |
| | w | 16.02 | 10.06 | 6.88 | 9.86 | 7.87 | 9.83 | 7.39 | 16.92 | 10.20 | 9.62 | 8.74 |
| $Ca^{2+}$ (cmol/Kg) | s | 8.88 | 9.01 | 7.75 | 5.38 | 11.5 | 6.13 | 4.13 | 4.75 | 10 | 5.5 | 4.63 |
| | w | 4.63 | 5.49 | 4.61 | 4.5 | 4.21 | 3.59 | 2.68 | 4.11 | 4.85 | 3.93 | 3.2 |
| $Mg^{2+}$ (cmol/Kg) | s | 6.38 | 5.97 | 4.32 | 2.37 | 4.73 | 2.88 | 1.85 | 2.26 | 3.5 | 2.67 | 2.67 |
| | w | 6.99 | 5.76 | 4.53 | 3.29 | 4.53 | 2.88 | 2.47 | 3.29 | 4.11 | 3.7 | 4.11 |
| $K^+$ (cmol/Kg) | s | 1.35 | 1.09 | 1.09 | 1.15 | 1.09 | 1.03 | 0.96 | 1.41 | 1.09 | 1.09 | 1.03 |
| | w | 0.77 | 1.28 | 0.77 | 0.64 | 0.64 | 0.9 | 0.64 | 1.03 | 0.77 | 0.51 | 0.77 |
| $Na^+$ (cmol/Kg) | s | 0.98 | 0.5 | 0.73 | 0.52 | 0.72 | 0.74 | 0.38 | 0.89 | 0.88 | 0.74 | 0.55 |
| | w | 0.65 | 0.43 | 0.54 | 0.54 | 0.43 | 0.54 | 0.54 | 0.76 | 1.09 | 0.87 | 0.54 |
| CEC (cmol/Kg) | s/w | 14.40 | 17.1 | 13.1 | 11.1 | 11.5 | 9.13 | 8.1 | 14 | 14.1 | 12.5 | 8.83 |
| V (%) | s | 122 | 96.8 | 106 | 84.6 | 156 | 118 | 90.3 | 66.8 | 109 | 80 | 101 |
| | w | 90.56 | 75.8 | 80 | 80.6 | 85.1 | 86.6 | 78.1 | 65.9 | 76.6 | 72.1 | 97.7 |
| SOM (%) | s | 5.49 | 5.87 | 5.28 | 2.91 | 4.26 | 2.2 | 2.1 | 3.32 | 2.5 | 2 | 0.6 |
| | w | 4.59 | 5.08 | 3.51 | 3.05 | 2.89 | 2.08 | 1.66 | 3.07 | 2.37 | 2.33 | 0.97 |
| N (%) | s | 0.28 | 0.17 | 0.19 | 0.15 | 0.13 | 0.12 | 0.08 | 0.15 | 0.14 | 0.12 | 0.05 |
| | w | 0.28 | 0.24 | 0.16 | 0.16 | 0.13 | 0.12 | 0.08 | 0.17 | 0.15 | 0.13 | 0.07 |
| C/N | s | 11.46 | 20.3 | 16.2 | 11.2 | 19 | 10.9 | 16.1 | 13 | 10.5 | 10 | 6.44 |
| | w | 9.62 | 12.5 | 12.7 | 10.8 | 12.9 | 9.96 | 12.1 | 10.4 | 9.46 | 10.3 | 8.09 |
| **Physical** | | | | | | | | | | | | |
| TC | s/w | SL | SL | SL | LS | LS | LS | LS | LS | SL | LS | LS |
| WHC (%) | s | 76.45 | 74.2 | 58 | 40.47 | 53.4 | 42.35 | 40.2 | 50.2 | 45.67 | 36.16 | 30.2 |
| | w | 56.43 | 63.9 | 49.2 | 45.5 | 45.5 | 39 | 36.8 | 44.9 | 45.8 | 40.5 | 35.9 |
| BD (g/cc) | s | 1.07 | 1.06 | 1.02 | 1.23 | 1.32 | 1.33 | 1.33 | 1.31 | 1.29 | 1.34 | 1.33 |
| | w | 1.13 | 1.00 | 0.97 | 1.17 | 1.24 | 1.16 | 1.28 | 1.06 | 1.16 | 1.22 | 1.42 |
| MI (Kpa) | s | 226 | 376 | 81 | 202 | 168 | 419 | 111 | 70 | 82 | 243 | 331 |
| | w | 162 | 220 | 126 | 227 | 196 | 430 | 178 | 121 | 131 | 181 | 287 |
| SD (cm) | s/w | 30 | 32.5 | 10 | 25 | 15 | 28 | 20 | 50 | 40 | 30 | 15 |

S: season (s: summer; w: winter); DCT: Dense Cork Trees, PTA: Pine Trees Afforestation, AOT: Abandoned Olive Trees, SCT: Sparse Cork Trees, DS: Dense Shrub, SS: Sparse Shrub, COT: Cultivated Olive Trees, BS: Bare soil, CV: Cultivated vineyards; EC: electrical conductivity; M: soil moisture content; $Ca^{2+}$, $Mg^{2+}$, $K^+$, $Na^+$ (exchangeable bases): calcium, magnesium, potassium and sodium; CEC: cationic exchange capacity; V: bases saturation; SOM: organic matter; N: total nitrogen; C/N: carbon-nitrogen ratio; TC: Textural class (SL: Sandy loam; LS: Loamy sand); WHC: water holding capacity; BD: bulk density; MI: mechanic impedance; SD: soil depth.

pH ranges from 6.0 to 6.8, a medium-neutral acid interval (Porta et al., 1994b), in both studied seasons and environments, with the exception of the cultivated vineyard environment (CV), with a pH of 5.2 (Norton et al., 1999). Soils with pH below 5.5, are considered to be acid, normally meaning that $Al^{3+}$ and $H^{+}$ ions displace $Ca^{2+}$ and $Mg^{2+}$ on the exchange complex enabling leaching to occur and decreasing the plant availability of some essential nutrients as well as altering many of the soil's chemical, physical and biological processes (Camberato, 2000; Sims, 2000). Hence, the CV seems to be at the first stage of an acidity problem, probably due to the cultivation and intense cropping combined with the low agricultural management (Norton et al., 1999; Dunjó et al., 2003). This could be a significant indicator of a slow but progressive acidification. However, further research is required for more reliable conclusions. With reference to the electrical conductivity (EC), results were not related to the land use and no significant differences were found between seasons. EC values were always lower than one dS/m and therefore no salinity problems were found (Carter, 1969; Martínez-Fernández et al., 1995).

Soil moisture content (M), seems slightly correlated with the land use and in particular with the land cover, and also being less in summer than in winter, due to drier meteorological conditions, as expected.

Generally, the cation exchange capacity (CEC), is dependent on the soil material and clay mineral (Sposito, 2000). There is not a clear pattern for the CEC in relation to the land-use sequence. In this context, environments with an early age of abandonment (A) present greater CEC values than those of later abandonment, such as land use B and C. However, this pattern is not accomplished for all the cultivated environments (D).

Base saturation (V) presents values greater than 60% in all environments and both seasons. However, an unbalanced percentage of exchangeable bases: calcium ($Ca^{2+}$), magnesium ($Mg^{2+}$), potassium ($K^{+}$), and sodium ($Na^{+}$), is revealed. According to Sims (2000), calcium should represent 20-80% of the CEC, being the dominant cation with a concentration in the soil solution higher than the rest of plant nutrients, whereas magnesium should occupy the 4-20%. Here, although the calcium is the dominant cation, there is a nutrient imbalance, mainly due to the deficiency of this exchangeable cation, which is four times lower than its optimal value in the relation $Ca^{2+}/Mg^{2+}$ and two times in relation to the $Ca^{2+}/K^{+}$ (Dunjó et al., 2003).

Soil organic matter (SOM) is considered the most important indicator of soil quality and therefore a key factor for its evaluation (Larson and Pierce, 1994; Acton and Gregorich, 1995; Sikora et al., 1996). In general, SOM data seem slightly higher in summer than in winter. A highly positive correlation is revealed between SOM content and the land use sequence in the whole study period. In this context, environments of early abandonment (A) such as cork trees (4.6-6%), and also pine tree afforestation (3.5-5.3%), have the highest SOM values. Then, SOM progressively decreases from the most recently abandoned (B,C) to the cultivated environments (D), where the cultivated vineyard presents the lowest value (0.6-1%) in both seasons because of its low management.

Similarly, the same pattern as SOM is shown by the total nitrogen (N) and water holding capacity (WHC), presumably accounting for the strong relationship between some of the most important soil properties and the land-use/land-cover type (Quiroga et al., 1998).

A C/N ratio value of 10 is believed the optimal for the best incorporation rate of the organic matter into the soil profile (Oades, 1984). Here, with the exception of the cultivated vineyard environment (CV), C/N ratio was always greater than 10 in all environments and seasons. However, the existence of hardly mineralizable decaying debris, has to be taken into account, which could account for the differences in the mineralization/humification rates among the selected environments, since it may entail a slower incorporation of the humic compounds into the soil profile, as well as the low total nitrogen content present in some environments (Dunjó et al., 2003).

In general soils of the study area are shallow (10-50 cm depth) and regolith often outcrops in most hilly eroded areas (Pardini et al., 2003). Environments are characterized by a similar particle size distribution, with the sand as the dominant fraction. Therefore, the textural class (TC) of these soils is sandy-loam or loamy-sand. Clay fraction was very low in all environments, probably reducing its contribution to the organo-mineral complex formation and enhancing the role of the humic faction alone in particle aggregation and water storage (Oades, 1984).

Bulk density (BD), is greater in summer than in winter for all environments, presumably due to different soil moisture conditions during the dry and wet season (Dunjó et al., 2003). This indicator of the soil compaction seems to be slightly correlated with land use, presenting lower values in the more vegetated environments (early age of abandonment), where soil organic matter content is higher, so that its effect on soil structure properties is more relevant (Dunjó et al., 2003). Mechanical impedance (MI) does not show any particular pattern in relation to the land use or the studied seasons.

#### 5.2.2.1. Analysis of variance for some of the main soil physico-chemical properties by land use and environment

Possible differences among the environments and/or land uses in the main soil-quality parameters were assessed through the analysis of variance (ANOVA) (Table 5.2).

Table 5.2. ANOVA test for assessing significant differences in means (for groups: land use type, and variables: environments) for statistical significance ($p<0.050$)

| **Independent variables** | **Dependent variables (N=4)** | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | **Water holding capacity** | | **Soil organic matter** | | **Nitrogen** | | **pH** | |
| | F-test | p-level | F-test | p-level | F-test | p-level | F-test | p-level |
| **Environments** | 10.551 | **0.000** | 12.523 | **0.000** | 6.461 | **0.000** | 13.826 | **0.000** |
| **Land use** | 20.265 | **0.000** | 26.558 | **0.000** | 9.387 | **0.000** | 0.455 | **0.715** |

The *dependent or quantitative variables* required for the ANOVA were soil quality parameters such as: soil organic matter (SOM), water holding capacity (WHC), soil total nitrogen (N) and pH. Data from these quantitative variables were from the whole study period, summer and winter seasons. On the other hand, the *independent or qualitative variables* were the eleven environments and the four land uses in which these were grouped (consult paper: Dunjó et al., 2003, for further details). ANOVA results were considered statistically significant when $p<0.05$. The results show significant differences between environments when assessing each soil-quality parameter ($p = 0.00$) (Table 5.2). This suggests that the selected environments are totally representative of the landscape diversity of the catchment. Likewise, there are significant differences among land-use groups in terms of soil-quality parameters, with the exception of the pH, which has a p value of 0.7. On the one hand, this result confirms that the comparative analysis among the four land uses is well founded, and on the other hand, it statistically justifies the previously mentioned assumption of the dependence of pH on parent material and explains why soils are characterised by acid conditions (Dunjó et al., 2003).

#### 5.2.2.2. Summary

It is suggested from the results that environments abandoned for the longest time have the best soil conditions. The lowest values of the main soil physico-chemical parameters, such as: SOM, N, WHC, CEC and pH, are associated to cultivated or recently abandoned environments (D) and especially to the cultivated vineyard environment (CV), whereas long term abandoned environments (A and also B), present better soil-quality properties. Therefore, it seems that the vegetation cover developed over time since abandonment might play an important role as a resilience factor to disturbances such as runoff generation, soil erosion and nutrient losses (Dunjó et al., 2003). The recovery of these ecosystems over a certain period of time, highlights their resilience to disturbances as a fundamental component of soil quality, since non-management activities are performed after farmland set-aside (Cammeraat and Imeson, 1998; Seybold et al., 1999). However, the more the soil has been neglected, the more its properties may be drastically altered (Lal, 1998), and therefore, concern about the threshold of resistance and resilience of these ecosystems stresses the necessity of gaining a better understanding of the overall dynamics of natural and anthropic disturbance (Pardini et al., 2002). Therefore, it is certain that soil agricultural management and soil condition, would be essential factors after land abandonment, for preserving the soil quality against the possibility of a slow but progressive degradation process (Pardini et al., 2002).

### 5.2.3. ASSESSMENT OF LAND-DEGRADATION PROCESSES

The protective effect of vegetation cover in degradation processes such as runoff or soil erosion has been widely studied in the Mediterranean region (e.g. Francis and Thornes, 1990; García-Ruíz et al., 1995). The canopy and litter vegetation covers act as direct mechanical protection of the soil surface, by intercepting rainfall and consequently reducing the detachment of soil particles caused by raindrop impact at the soil surface (Elwell and Stocking, 1976; Bochet et al., 1998; Bochet et al., 1995). On the other hand, soil physico-chemical properties are indirectly improved through the incorporation of the organic matter from the vegetation (Boix-Fayos et al., 1998). Furthermore, a part from the vegetation-cover type and percentage, the soil type, land use and rainfall erosivity are expected to have an important role in hydrological and soil erosion processes (Wischmeier et al., 1971; Luk, 1979; Lal, 1988b; Bajaracharya and Lal, 1992; Romero-Díaz et al., 1999).

#### 5.2.3.1. Characteristics of the runoff-erosion microplots

The vegetation cover of each runoff-erosion microplot was determined in three different ways: (a) the undergrowth plant canopy (UPC), or the ground vegetation covering the microplot; (b) the aerial plant canopy (APC), or arboreal vegetation coving the microplot; and (c) the total vegetation cover (VC), which accounts for both, the UPC and the APC (Table 5.3).

Table 5.3. Main microplot field characteristics related to each selected environment

| Environments | | Land use-cover | H (m) | S (%) | A | ST (%) | BS (%) | UPC (%) | AC (%) | VC (%) |
|---|---|---|---|---|---|---|---|---|---|---|
| Dense Cork Trees | (DCT) | A | 65 | 25 | E-W | 5 | 10 | 90 | 80 | 85 |
| Dense Cork Trees | (DCT) | (50 years) | 180 | 22 | E-W | 10 | 15 | 75 | 65 | 75 |
| Pine Trees Afforestation | (PTA) | | 235 | 25 | N-S | 15 | 5 | 82 | 70 | 80 |
| Abandoned Olive Trees | (AOT) | B | 155 | 20 | W-E | 5 | 20 | 72 | 82 | 75 |
| Sparse Cork Trees | (SCT) | (25 years) | 225 | 24 | N-S | 50 | 20 | 32 | 0 | 30 |
| Dense Shrub | (DS) | C | 210 | 18 | N-S | 20 | 15 | 67 | 0 | 65 |
| Dense Shrub | (DS) | (5 years) | 160 | 18 | W-E | 10 | 15 | 77 | 0 | 75 |
| Sparse Shrub | (SS) | | 200 | 18 | E-W | 25 | 20 | 55 | 0 | 55 |
| Cultivated Olive Trees | (COT) | D | 95 | 25 | N-S | 5 | 70 | 10 | 55 | 25 |
| Bare Soil | (BS) | (0 years) | 165 | 20 | N-S | 20 | 50 | 30 | 0 | 30 |
| Cultivated Vineyards | (CV) | | 55 | 18 | W-E | 15 | 70 | 12 | 0 | 15 |

H: altitude; S: slope; A: slope aspect; ST: stoniness; BS: bare soil; UPC: undergrowth plant canopy; AC: aerial plant canopy; VC: total vegetation cover. Mean data of both environment microplots.

Moreover, the percentage of bare soil (BS) and stoniness (ST) was also determined for each microplot (Table 5.3). The altitude (H), slope (S) and orientation or aspect (A) of each microplot was also assessed (Table 5.3). Data in Table 5.3 are expressed for each environment and correspond to the mean value of both microplot determinations in that environment.

### 5.2.3.2. Rainfall data

Monthly rainfall amount and intensity (year: 1999 and 2000), provided by the Figueres automatic weather station (AWS) are illustrated in Figure 5.1.

The recorded data show a high inter-seasonal as well as inter-annual rainfall variability in the study area. Assessing both years, the greatest recorded rainfall amount was in September (124 l/m$^2$) and December (134.1 l/m$^2$), respectively; whereas the lowest was in February (0.6 l/m$^2$ in 1999 and 11.9 l/m$^2$ in 2000).

In addition to the AWS, a set of rain gauges from 5 to 10 litre volume were randomly set up in the catchment. Nine rainfall events were recorded in the study area, five in summer (max: 35 l/m$^2$; min: 5.5 l/m$^2$) and four in winter (max: 50 l/m$^2$; min: 9 l/m$^2$) (Table 5.4). Contrary to the seasonal precipitation pattern typical of the Mediterranean region, the highest total rainfall amount was unexpectedly revealed in summer, during the driest season (137 l/m$^2$) (Table 5.4).

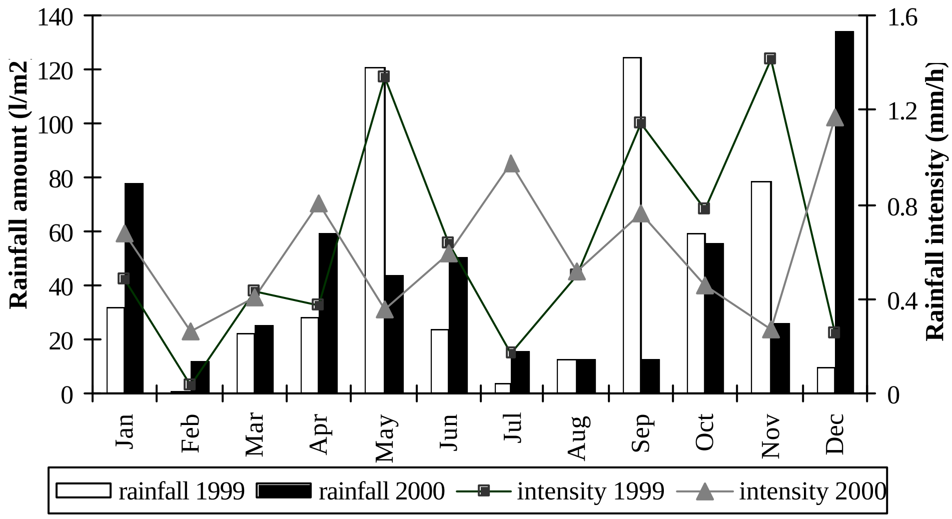


Figure 5.1. Monthly rainfall amount and intensity over the study period and area

Table 5.4. Rainfall amounts for individual rainfall events and study season

| | Summer 1999 | | | | | Winter 2000 | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Date** | **25-08** | **31-08** | **07-09** | **16-09** | **23-09** | **21-01** | **25-02** | **03-03** | **28-03** |
| **Rainfall (l/m$^2$)** | 6 | 10 | 78 | 35 | 5,5 | 50 | 9 | 26 | 20 |
| **Total Rainfall (l/m$^2$)** | | | 137 | | | | 105 | | |

### 5.2.3.3. Rainfall-runoff-erosion relationships

In most cases, runoff-erosion microplots show a high variability in terms of response, to the nine recorded rainfall events (Dunjó et al., in press). The minimum rainfall amount required for generating runoff and soil loss in summer and in all environments was approximately 72 l/m$^2$ in both cases, whereas about 50 l/m$^2$ was needed for sediment yield in winter (Table 5.5 and 5.6). Regarding the highest rainfall event in winter (50 l/m$^2$), only six of the eleven assessed environments presented significant amount of runoff generation (Table 5.5 and 5.6). Thus, winter results suggest that higher rainfall amounts are required for generating runoff in all the environments, stressing the importance of soil moisture conditions before rainfall. Although, runoff and erosion rates should be lower in summer than in winter, accounting for the drier soil conditions which entail better soil infiltration capacity and consequently lower runoff and sediment yield, these processes showed their highest response in summer for all environments.

Table 5.5. Summer daily runoff and sediment yield measurements per environment

| | | Selected Environments | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Variable** | **Date** | **DCT** | **COT** | **BS** | **AOT** | **DS** | **SS** | **DCT** | **SCT** | **PTA** | **CV** | **DS** |
| **Runoff (l/m2)** | 25-08-99 | nd | 0.59 | 0.71 | nd | nd | 0.67 | 0.05 | 0.09 | 0.06 | nd | nd |
| | 31-08-99 | 0.03 | 0.6 | 0.17 | 0.07 | 0.02 | nd | 0.04 | 0.11 | nd | 1.09 | nd |
| | 07-09-99 | 7.03 | 6.59 | 2.39 | 5.04 | 0.58 | 6.1 | 1.27 | 12.09 | 2.34 | 4.44 | 2.97 |
| | 16-09-99 | 0.12 | 13.49 | 2.99 | 1.31 | 0.3 | 11.11 | nd | 9.44 | 8.02 | 4.24 | 2.18 |
| | 23-09-99 | nd | 0.17 | 0.17 | 0.01 | nd | nd | nd | nd | 0.22 | 1.58 | 0.02 |
| **Eroded Soil (g/m2)** | 25-08-99 | nd | 20.13 | 7.08 | nd | nd | 1.98 | nd | 6.21 | 1.47 | nd | nd |
| | 31-08-99 | 0.14 | 11.26 | 5.74 | 0.83 | 2.32 | 1.46 | nd | 0.28 | 0.37 | 7.61 | 0.47 |
| | 07-09-99 | 3.19 | 648.43 | 82.35 | 13.25 | 7.15 | 4.73 | 0.14 | 22.85 | 67.13 | 40.93 | 14.77 |
| | 16-09-99 | 1.21 | 96.19 | 18.33 | 3.69 | 6.41 | 11.68 | nd | 10.72 | 41.53 | 9.04 | 5.77 |
| | 23-09-99 | 0.32 | 21.35 | 0.54 | 0.26 | 0.57 | 0.77 | nd | nd | 1.88 | 1.86 | 0.61 |

DCT: Dense Cork Trees, PTA: Pine Trees Afforestation, AOT: Abandoned Olive Trees, SCT: Sparse Cork Trees, DS: Dense Shrub, SS: Sparse Shrub, COT: Cultivated Olive Trees, BS: Bare soil, CV: Cultivated vineyards; nd: not found/ not analysed.

Therefore, it seems that in this area rainfall amount and particularly intensity are key factors of the land-degradation processes. However, it has been observed that the heaviest storms do not necessarily coincide with the greatest rates of runoff and sediment, as for example those related to the event of 78 l/m$^2$ and 50 l/m$^2$ for summer and winter respectively (Table 5.4, 5.5 and 5.6).

Moreover, it is observed that adjacent plots may present a high variability in runoff and sediment yields for the same rainfall event. Runoff levels oscillate between 0.02 and 13.49 l/m$^2$, whilst sediment yields range from 0.12-96.19 g/m$^2$ over both seasons and all environments, with the exception of the cultivated olive trees environment where

648.43 g/m$^2$ were measured after the greatest rainfall event (78 l/m$^2$) in summer (Table 5.5 and 5.6).

Table 5.6. Winter daily runoff and sediment yield measurements per environment

| | | Selected Environments | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Variable** | **Date** | **DCT** | **COT** | **BS** | **AOT** | **DS** | **SS** | **DCT** | **SCT** | **PTA** | **CV** | **DS** |
| **Runoff (l/m2)** | 21-01-00 | nd | 10.88 | 0.85 | nd | nd | 0.92 | nd | 0.87 | 2.37 | 2.29 | nd |
| | 25-02-00 | nd | 0.84 | nd | nd | nd | 0.11 | nd | nd | nd | nd | nd |
| | 03-03-00 | nd | nd | 0.05 | 0.28 | nd | nd | nd | 0.57 | 0.39 | nd | 0.05 |
| | 28-03-00 | nd | 1.72 | 0.14 | nd | nd | 0.99 | nd | 0.32 | 0.61 | 1.71 | nd |
| **Eroded Soil (g/m2)** | 21-01-00 | 3.73 | 22.37 | 0.81 | 1.49 | 1.45 | 5.22 | 0.55 | 1.99 | 5.94 | 8.32 | 0.94 |
| | 25-02-00 | nd | 50.97 | nd | nd | nd | 4.16 | nd | nd | nd | 0.76 | nd |
| | 03-03-00 | 0.94 | 30.98 | 0.83 | nd | nd | nd | nd | nd | 1.45 | 1.57 | nd |
| | 28-03-00 | 0.15 | 22.91 | 0.01 | nd | 0.23 | 0.63 | 0.06 | 0.12 | 1.32 | 3.34 | nd |

DCT: Dense Cork Trees, PTA: Pine Trees Afforestation, AOT: Abandoned Olive Trees, SCT: Sparse Cork Trees, DS: Dense Shrub, SS: Sparse Shrub, COT: Cultivated Olive Trees, BS: Bare soil, CV: Cultivated vineyards; nd: not found/ not analysed.

Erosion rates and runoff production in the eleven environments are considered low over the study period and particularly in winter. On the one hand, these low rates may presumably be due to the lower recorded rainfall amount and intensity (winter), and on the other hand, it may be connected to the design of the bounded plots. These plots are disconnected from the natural hillslope, forming a closed system where sediment transported downslope is not replaced by upslope material, with the result that the more readily erodible material tends to become depleted with the time (Vacca et al., 2000; Romero-Díaz et al., 1999).

The analysis of the results of individual events from August-September 1999 and January-March 2000, was performed using a correlation matrix of Pearson coefficients (Table 5.7). This study reveals a direct correlation between rainfall and runoff and soil loss, when evaluating both seasons separately as well as the whole study period (P-R: +0.51; P-ES: +0.31) (Table 5.7). In this context, runoff and eroded soil are significantly correlated in a positive way (s: +0.30; w: +0.38; s-w: +0.31). Vegetation cover (VC) presents a reasonable negative relationship with bare soil (-0,86), stone cover (-0,41) an runoff (-0,24), although no significant correlation has been found with the eroded soil (-0,19). Correlation coefficients from all the analysed variables are slightly higher in winter than in summer.

Table 5.7. Correlation matrix by Pearson coefficients, among sediment yield, rainfall, runoff, vegetation cover, stone cover and bare soil for both seasons separately (summer: N=55 and winter: N=44) as well as the entire study period (N=99)

| | Season | Eroded Soil | Rainfall | Runoff | Vegetation | Stoniness | Bare Soil |
|---|---|---|---|---|---|---|---|
| **Eroded Soil** | s | | | | | | |
| | w | 1,00 | | | | | |
| | s-w | | | | | | |
| **Rainfall** | s | 0,34 | | | | | |
| | w | 0,02 | 1,00 | | | | |
| | s-w | 0,31 | | | | | |
| **Runoff** | s | 0,30 | 0,57 | | | | |
| | w | 0,38 | 0,34 | 1,00 | | | |
| | s-w | 0,31 | 0,51 | | | | |
| **Vegetation** | s | -0,23 | -0,00 | -0,24 | | | |
| | w | -0,40 | -0,00 | -0,31 | 1,00 | | |
| | s-w | -0,19 | -0,00 | -0,24 | | | |
| **Stoniness** | s | -0,13 | -0,00 | 0,19 | -0,41 | | |
| | w | -0,26 | 0,00 | -0,09 | -0,41 | 1,00 | |
| | s-w | -0,11 | -0,00 | 0,10 | -0,41 | | |
| **Bare Soil** | s | 0,32 | 0,00 | 0,16 | -0,86 | -0,11 | |
| | w | 0,58 | -0,00 | 0,38 | -0,86 | -0,11 | 1,00 |
| | s-w | 0,27 | 0,00 | 0,21 | -0,86 | -0,11 | |

s: summer; w: winter; s-w: summer and winter. Numbers in bold are significant at the p< 0,05 level. Plus sign (+) refers to positive correlation and minus sign (-) to negative correlation.

### 5.2.3.4. Assessment of the effect of land-use/cover on runoff generation and sediment yield

Rainfall-runoff and rainfall-soil loss relationships were statistically assessed for each environment, land use and for both studied seasons, through the performance of linear correlations (Figure 5.2 and 5.3 respectively). The linear regression equations as well as the associated correlation coefficient, $R^2$, are presented in Table 5.8.

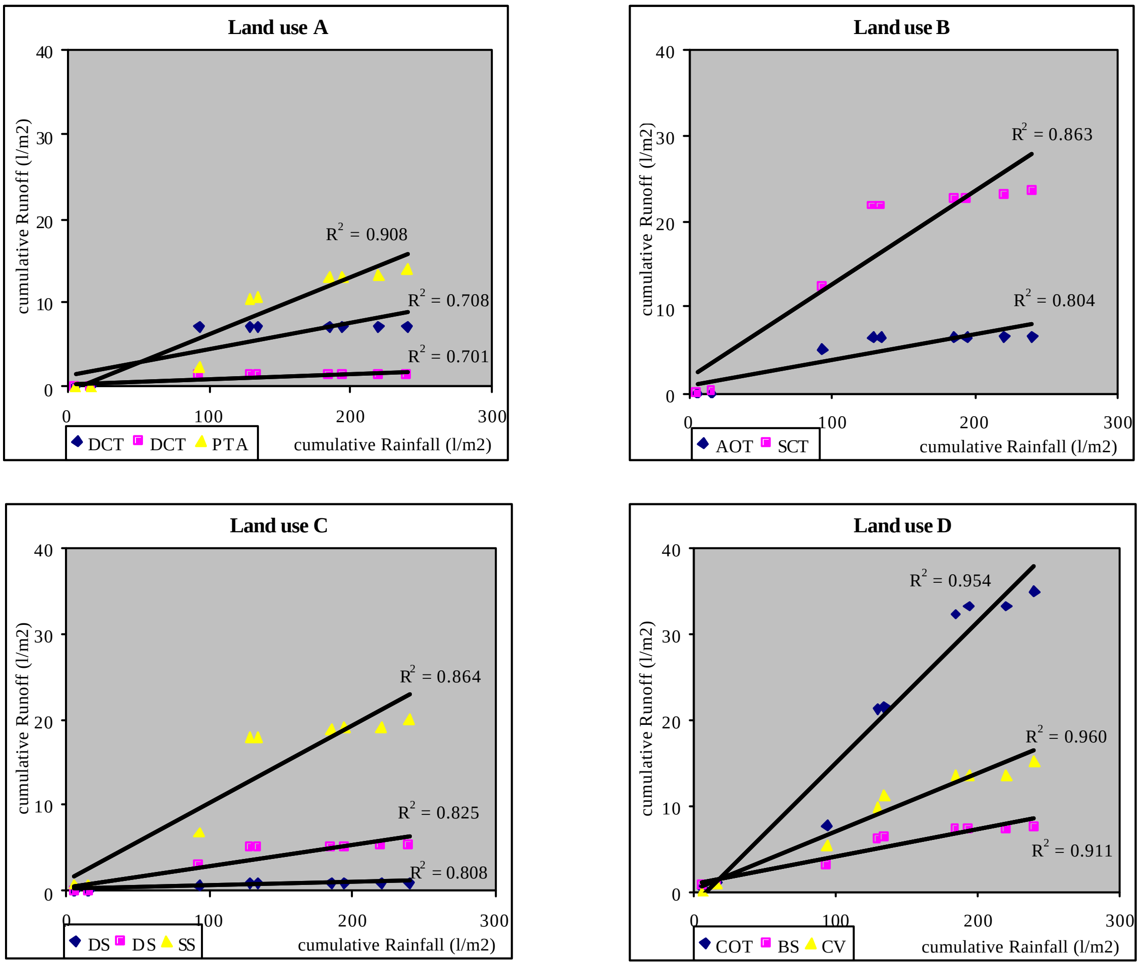


Figure 5.2. Cumulative runoff versus cumulative rainfall amount per land use-cover from summer and winter data

Best rainfall-runoff correlations ($R^2 > 0.9$) were associated to land use D (CV, COT and BS), and to the pine trees afforestation (land use A) (Figure 5.2). The highest runoff rate was assessed in the cultivated olive trees environment, for the overall period. However, rainfall and runoff were highly positively correlated in all environments ($0.7 < R^2 > 0.9$) and in all land uses, the two cork trees environments (land use A) showing the least correlation (Figure 5.2; Table 5.8). With reference to rainfall-soil loss, all environments showed similar positive excellent correlations ($0.78 < R^2 > 0.98$) (Figure 5.3; Table 5.8). There were no significant differences between environments nor land uses.

However, in terms of amount of erosion per unit rainfall, the cultivated or early age of abandonment environments (land use D) had the greatest soil loss rates, and in particular the cultivated olive trees (COT), with a cumulative sediment yield of: 924.59 g/m$^2$, for the whole studied period (Table 5.9). Thus, soil loss rates seem to experience a progressive decline from land use D environments to the pine trees afforestation (land use A), land covers B, C and finally the cork-tree environments (land use A), throughout the study period (Table 5.9).

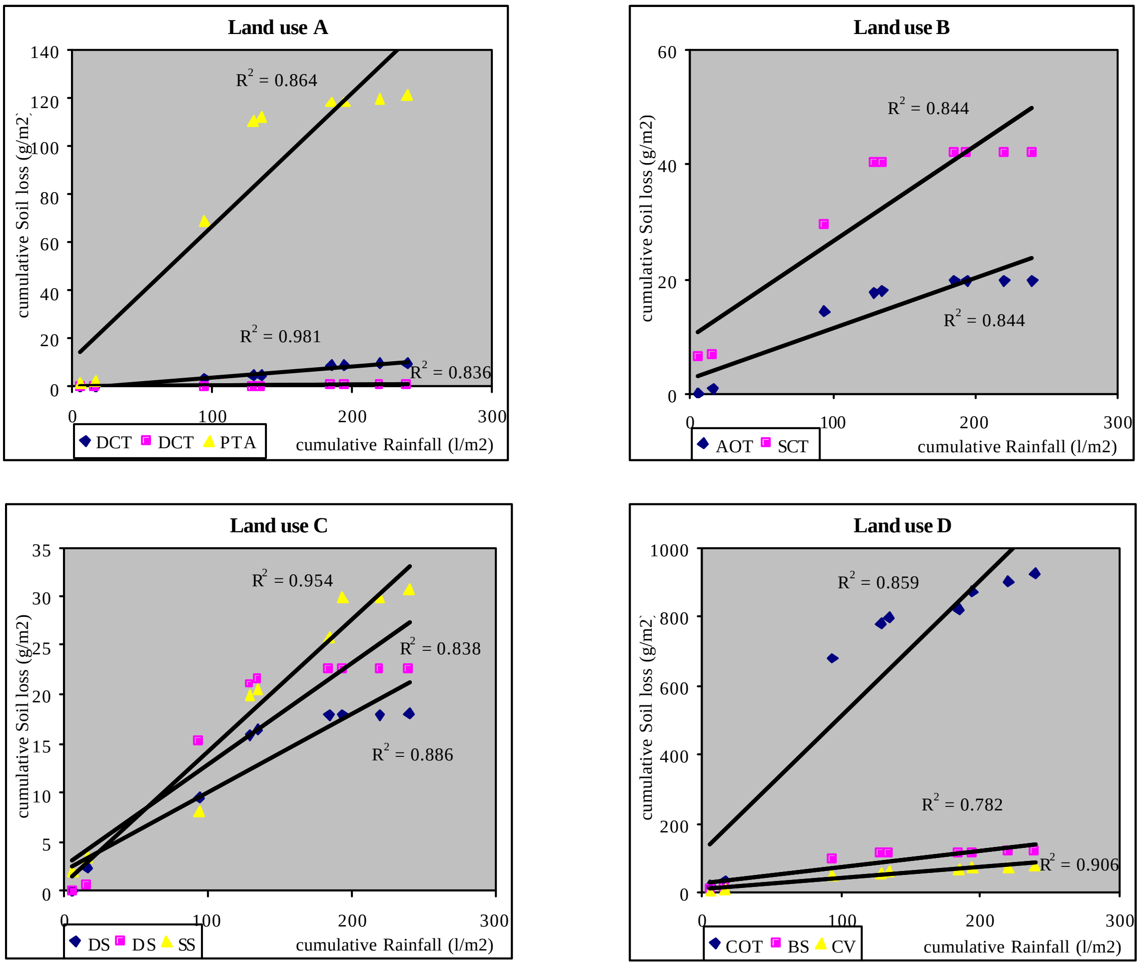


Figure 5.3. Cumulative soil loss versus cumulative rainfall amount per land use-cover from summer and winter data

The highest rates of runoff and soil loss correspond to the land use D environments, particularly the cultivated olive trees (COT), with a slope of 25%, a bare soil cover of 70% and only 25% of vegetation cover, and thus the environment with the greatest potential to be degraded, since it has the highest runoff and sediment yield, in both studied seasons (Dunjó et al., in press) (Table 5.9). Although the vegetation cover and slope angle are very similar in the land use A environments, the pine trees afforestation showed significantly higher rates of both soil erosion and runoff, presumably accounting for the hardly pine mineralizable decaying debris. According to Zanchi (1993), these measurements of runoff and soil erosion rates are considered low, even the ones from the cork trees environment.

Table 5.8. Linear regressions and correlation coefficients ($R^2$) for runoff and soil loss vs. rainfall in the eleven environments and both studied seasons

| | | Linear correlations | | | |
|---|---|---|---|---|---|
| **Environment** | **Land use** | **Soil loss vs Rainfall** | **$R^2$** | **Runoff vs Rainfall** | **$R^2$** |
| **DCT** | **A** | y = 0.0451x - 0.6315 | 0.981 | y = 0.0315x + 1.312 | 0.708 |
| **DCT** | **A** | y = 0.0036x - 0.128 | 0.836 | y = 0.0056x + 0.3028 | 0.701 |
| **PTA** | **A** | y = 0.5559x + 10.67 | 0.864 | y = 0.0673x - 0.5509 | 0.908 |
| **SCT** | **B** | y = 0.1663x + 9.7819 | 0.844 | y = 0.1086x + 1.738 | 0.863 |
| **AOT** | **B** | y = 0.088x + 2.4045 | 0.844 | y = 0.03x + 0.8704 | 0.804 |
| **DS** | **C** | y = 0.0804x + 2.0046 | 0.886 | y = 0.0244x + 0.4803 | 0.825 |
| **DS** | **C** | y = 0.1035x + 2.5161 | 0.838 | y = 0.004x + 0.1091 | 0.808 |
| **SS** | **C** | y = 0.1346x + 0.7482 | 0.954 | y = 0.0907x + 1.1081 | 0.864 |
| **COT** | **D** | y = 3.9577x + 111.55 | 0.859 | y = 0.1641x - 1.5508 | 0.954 |
| **CV** | **D** | y = 0.3111x + 8.2561 | 0.906 | y = 0.0671x + 0.2649 | 0.960 |
| **BS** | **D** | y = 0.4777x + 24.686 | 0.783 | y = 0.0319x + 0.8984 | 0.911 |

DCT: Dense Cork Trees, PTA: Pine Trees Afforestation, AOT: Abandoned Olive Trees, SCT: Sparse Cork Trees, DS: Dense Shrub, SS: Sparse Shrub, COT: Cultivated Olive Trees, BS: Bare soil, CV: Cultivated vineyards; $R^2$: correlation coefficient.

In general, the greatest runoff and erosion rates are related to cultivated or recently abandoned environments (D), whereas the lowest to the long term abandoned environments (A) with the exception of the pine tree afforestation environment (Figure 5.4). However it cannot be said that a clear pattern exists in the runoff and sediment yield neither among the land use sequence nor the environments.

Table 5.9. Cumulative soil loss and runoff per environment, season and overall period

| Variable | | Land use A, B, C, D and selected Environments | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | A | | B | | | C | | | D | |
| | S | DCT | COT | PTA | AOT | SCT | DS | DS | SS | COT | BS | CV |
| ES | s | 4.9 | 0.1 | 112.4 | 18.0 | 40.1 | 16.5 | 21.6 | 20.6 | 797.4 | 114.0 | 59.44 |
| ($g/m^2$) | w | 4.8 | 0.6 | 8.7 | 1.5 | 2.1 | 1.7 | 0.9 | 10.0 | 127.2 | 1.7 | 14.0 |
| | s-w | 9.7 | 0.8 | 121.1 | 19.5 | 42.2 | 18.1 | 22.6 | 30.6 | 924.6 | 115.7 | 73.4 |
| R | s | 7.2 | 1.3 | 10.6 | 6.4 | 21.7 | 0.9 | 5.2 | 17.9 | 21.4 | 6.4 | 11.4 |
| ($l/m^2$) | w | 0.0 | 0.0 | 3.4 | 0.3 | 1.8 | 0.0 | 0.1 | 2.0 | 13.4 | 1.0 | 4.0 |
| | s-w | 7.2 | 1.3 | 14.0 | 6.7 | 23.5 | 0.9 | 5.2 | 19.9 | 34.9 | 7.5 | 15.4 |

ES: soil loss; R: runoff generation; S: season (s: summer; w: winter); DCT: Dense Cork Trees, PTA: Pine Trees Afforestation, AOT: Abandoned Olive Trees, SCT: Sparse Cork Trees, DS: Dense Shrub, SS: Sparse Shrub, COT: Cultivated Olive Trees, BS: Bare soil, CV: Cultivated vineyards.

Runoff and soil-erosion data from summer and winter separately, per environment and land use are presented in Figure 5.4, where values of both parameters are clearly higher in summer, since more rainfall events were recorded in this drier season (summer).

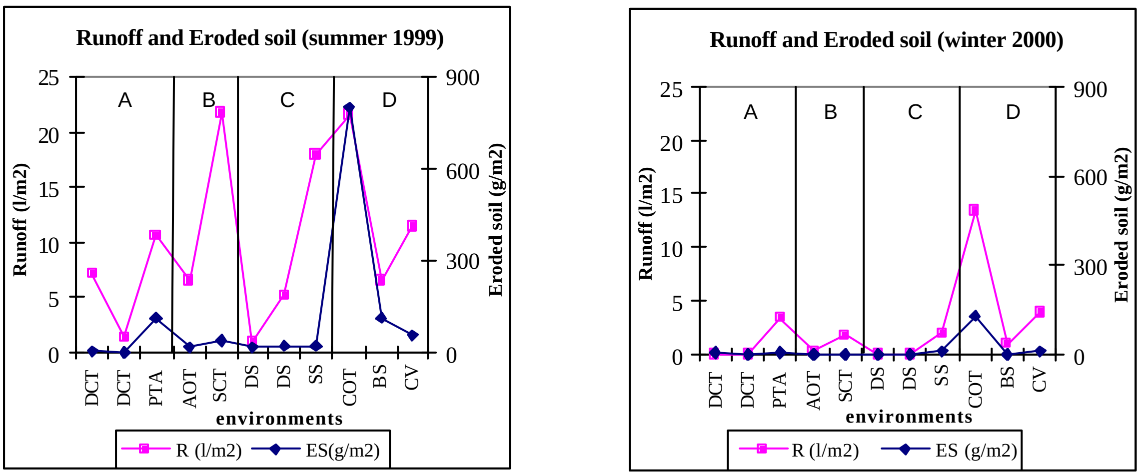


Figure 5.4. Cumulative runoff and sediment yield per environment, land use and season

The cultivated olive trees environment (COT), with a 85% bare soil and a slope of 24%, showed the highest rates of runoff and erosion in both studied seasons. With 70% of bare soil and the lowest slope percentage (11%), the cultivated vineyard also from the land use D, was the second environment with highest rates of runoff and soil loss. The third, was the pine trees afforestation environment (PTA), with the steepest slope

(47%), but the lowest bare soil percentage (5%), and 70 and 82% of aerial and undergrowth plant canopy respectively. This latter example, accounts for the potential fragility in terms of physical and chemical degradation of those well-vegetated environments without any conservation management, after a reforestation strategy (Dunjó et al., 2003). The lowest rates of both runoff and eroded soil were measured in the dense shrub environments (land use C), which presented a high percentage of undergrowth plant canopy, indicating the important role of vegetation cover as a soil protector.

#### 5.2.3.5. Analysis of variance for runoff and erosion by season, environment and land use

The analysis of variance (ANOVA) was performed for contrasting differences in sediment yield and runoff rates across land uses and selected environments (Thompson and Robert, 1995) (Table 5.10). In the analysis, runoff and soil erosion data were the measured or dependent variables, whereas the selected environments and the land uses were the controlled or independent variables. Three ANOVA tests were conducted, two for each of the studied seasons and another per the whole period (Table 5.10).

Table 5.10. Results of ANOVA used to test soil erosion and runoff, for differences in means (for groups: land use; and variables: environments)

| Dependent Variables | Independent variables | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | | Environments (1-11) | | Land use-cover (A, B, C, D) | | | |
| | S | N | F-test | p-level | F-test | p-level | Mean | Std dev |
| **Eroded Soil** | s | **55** | **1,504** | **0,170** | **1,683** | **0,182** | **21,909** | **88,291** |
| **Runoff** | s | **55** | **0,944** | **0,503** | **0,643** | **0,591** | **2,009** | **3,386** |
| **Eroded Soil** | w | **44** | **18,241** | 0,000 | **4,602** | 0,007 | **3,937** | **9,758** |
| **Runoff** | w | **44** | **1,485** | **0,189** | **1,815** | **0,160** | **0,590** | **1,704** |
| **Eroded Soil** | s-w | **99** | **1,993** | 0,043 | **2,172** | **0,096** | **13,922** | **66,466** |
| **Runoff** | s-w | **99** | **1,578** | **0,126** | **1,234** | **0,302** | **1,378** | **2,845** |

S: season (s: summer; w: winter); N: number of assessed samples; Std dev: standard deviation. Significant differences in bold: p<0.050.

Significant differences ($p<0.050$) in soil loss among the eleven environments ($p=0.000$), and the four land use-cover types ($p=0.007$), were revealed in the winter season, and only among the environments when assessing the overall period ($p=0.043$) (Table 5.10). No significant results were found with the runoff data, probably because of its high variability (Dunjó et al., in press).

#### 5.2.3.6. Summary

In the Serra de Rodes catchment, reforested environments left without management, such as the pine trees afforestation (PTA), seem to be going through a more intense soil degradation than those environments left under natural vegetation (land use C) (Aru and Barrocu, 1993). Data analysis reveals the cultivated or recently abandoned environments (land use D), as the most likely to be degraded in terms of runoff generation and soil erosion. In particular, the highest rates of runoff and soil loss were associated with the cultivated olive trees environment (COT). Chemical pesticides are still applied to few remaining cultivated olive trees patches (COT) mostly located in the floodplain, entailing the total removal of the soil vegetation and consequently increasing the runoff and soil-erosion rates of these environments (Dunjó et al., in press). However, olives in the study area generally grow under semi-natural or even abandoned conditions (AOT), mainly due to the high competition of olive oil with other crops producing higher incomes (Dunjó et al., 2003). Therefore, the presence of annual vegetation and plant residues on the soil surface effectively prevents the surface sealing formation and could drastically reduces the runoff water velocity and soil loss. These environments could provide some resistance to further degradation processes in hilly areas. In general, Mediterranean soils cultivated with vines typically remain almost bare during autumn, winter and early spring due to removal of annual vegetation by ploughing or application of pesticides (Kosmas et al., 1997). Regarding the cultivated vineyard (CV) and bare soil (BS) environments (land use D), both also showed high rates of runoff and soil loss in comparison to the remaining land uses and environments, because of the fragility of soils under null or very low agricultural management

(Romero-Díaz et al., 1999). The general pattern from runoff-erosion microplots measurements, in relation to land use C environments, revealed the sparse shrubs as more likely to be degraded than those better-vegetated areas in agreement with Nicolau et al.(1996).

### 5.2.4. SOIL-QUALITY INDICES

The aim of performing a principal components analysis (PCA) was to simplify and summarize the contribution of the overall set of measured soil physico-chemical variables, by grouping them into several factors or indices.

Table 5.11. Results from the principal components analysis of the minimum rainfall volume recorded from summer (5.5 l/m$^2$) and winter (9.0 l/m$^2$)

| Variables | FI | | FII | | FIII | |
|---|---|---|---|---|---|---|
| | s | w | s | w | s | W |
| Runoff volume (l/m$^2$) | **+0.758** | **+0.986** | -0.323 | -0.011 | +0.031 | -0.002 |
| Dissolved calcium (mg/m$^2$) | **+0.917** | **+0.986** | +0.211 | -0.011 | +0.070 | -0.002 |
| Dissolved magnesium (mg/m$^2$) | **+0.913** | **+0.985** | +0.073 | +0.045 | +0.019 | +0.009 |
| Dissolved potassium (mg/m$^2$) | **+0.618** | **+0.987** | -0.171 | +0.003 | -0.169 | +0.000 |
| Dissolved sodium (mg/m$^2$) | **+0.915** | **+0.985** | -0.133 | +0.047 | +0.024 | +0.009 |
| Dissolved organic carbon (mg/m$^2$) | **+0.934** | **+0.986** | +0.101 | -0.015 | +0.090 | -0.003 |
| Dissolved nitrogen (mg/m$^2$) | **+0.879** | **+0.987** | +0.127 | +0.026 | +0.008 | +0.005 |
| Eroded soil (g/m$^2$) | **+0.936** | **+0.987** | -0.236 | +0.011 | -0.024 | +0.011 |
| Eroded organic carbon (g/m$^2$) | **+0.979** | **+0.985** | -0.084 | -0.022 | -0.047 | +0.000 |
| Eroded nitrogen (g/m$^2$) | **+0.940** | **+0.987** | -0.012 | -0.004 | -0.019 | +0.006 |
| Vegetal debris (g/m$^2$) | **+0.648** | **+0.986** | +0.524 | -0.015 | +0.285 | +0.003 |
| Aerial canopy (%) | +0.102 | +0.248 | **+0.843** | **+0.722** | -0.114 | -0.255 |
| Water holding capacity (%) | -0.338 | -0.050 | **+0.793** | **+0.962** | +0.360 | -0.142 |
| Sand fraction (%) | +0.213 | +0.083 | **-0.777** | **-0.764** | +0.094 | +0.370 |
| Clay fraction (%) | +0.363 | +0.477 | **+0.669** | **+0.483** | +0.199 | -0.254 |
| Cation exchange capacity (cmol/Kg) | +0.043 | +0.168 | **+0.825** | **+0.916** | +0.071 | -0.034 |
| Soil organic matter (%) | -0.456 | -0.175 | **+0.821** | **+0.929** | +0.218 | -0.198 |
| Soil total nitrogen (%) | -0.209 | -0.044 | **+0.830** | **+0.940** | +0.387 | +0.038 |
| Slope (%) | --- | +0.083 | --- | +0.418 | **---** | **-0.740** |
| Undergrowth plant canopy (%) | -0.397 | -0.397 | +0.208 | +0.070 | **-0.705** | **-0.525** |
| Bare soil (%) | --- | +0.532 | --- | +0.134 | **---** | **+0.602** |
| Absolute total variance (%) | 32.84 | 42.22 | 25.55 | 26.64 | 9.34 | 8.64 |
| Accumulate total variance (%) | 32.84 | 42.22 | 58.39 | 68.86 | 67.73 | 77.51 |

Three principal components were identified and variables grouped to FI, FII and FIII. Percentages of the explained variability and sense of correlation (+ or -) of each variable in each component and percentages of the explained total variability of each variable by the three components are presented.

PCA assessed possible correlations among variables by using the minimum number of factors to describe them (Paniagua et al., 1999). Two PCA (four routines) were carried out for the maximum and minimum rainfall events recorded during both studied seasons (Table 5.11 and 5.12 respectively). Three principal components were identified and significant variables grouped accordingly. The first factor (FI) addressed soil erosion, runoff generation and nutrient losses, the second (FII) was related to soil physico-chemical properties and accounted for soil quality, and the third component (FIII) comprised variables related to soil protection (Dunjó et al., 2003). Further information about the PCA methodology, used can be consulted in Dunjó et al. (2003).

Table 5.12. Results from the principal factor analysis of the maximum rainfall volume recorded from summer (35.0 l/m$^2$) and winter (50.0 l/m$^2$)

| **Variables** | **FI** | | **FII** | | **FIII** | |
|---|---|---|---|---|---|---|
| | **s** | **W** | **s** | **w** | **s** | **w** |
| Runoff volume (l/m$^2$) | **+0.909** | **+0.983** | -0.116 | -0.022 | -0.120 | +0.044 |
| Dissolved calcium (mg/m$^2$) | **+0.852** | **+0.961** | -0.018 | -0.0138 | +0.462 | +0.158 |
| Dissolved magnesium (mg/m$^2$) | **+0.957** | **+0.969** | -0.060 | -0.000 | +0.130 | +0.123 |
| Dissolved potassium (mg/m$^2$) | **+0.899** | **+0.959** | +0.019 | +0.057 | -0.049 | +0.147 |
| Dissolved sodium (mg/m$^2$) | **+0.681** | **+0.834** | -0.337 | -0.008 | -0.096 | -0.411 |
| Dissolved organic carbon (mg/m$^2$) | **+0.798** | **+0.967** | +0.040 | -0.065 | +0.480 | +0.141 |
| Dissolved nitrogen (mg/m$^2$) | **+0.834** | **+0.949** | +0.070 | -0.045 | +0.442 | +0.105 |
| Eroded soil (g/m$^2$) | **+0.889** | **+0.972** | +0.113 | -0.066 | +0.346 | +0.093 |
| Eroded organic carbon (g/m$^2$) | **+0.887** | **+0.917** | +0.126 | +0.095 | +0.373 | -0.125 |
| Eroded nitrogen (g/m$^2$) | **+0.600** | **+0.871** | -0.051 | -0.145 | -0.253 | +0.189 |
| Aerial plant canopy (%) | +0.021 | +0.251 | **+0.714** | **+0.772** | +0.309 | -0.173 |
| Water holding capacity (%) | -0.248 | -0.083 | **+0.888** | **+0.958** | +0.115 | -0.152 |
| Sand fraction (%) | +0.017 | +0.012 | **-0.857** | **-0.687** | +0.206 | +0.595 |
| Clay fraction (%) | +0.243 | +0.518 | **+0.544** | **+0.534** | +0.520 | -0.074 |
| Cation exchange capacity (cmol/Kg) | -0.052 | +0.100 | **+0.761** | **+0.917** | +0.333 | -0.001 |
| Soil organic matter (%) | -0.149 | -0.221 | **+0.948** | **+0.943** | -0.094 | -0.159 |
| Soil total nitrogen (%) | -0.217 | -0.122 | **+0.820** | **+0.963** | +0.382 | +0.010 |
| Bulk density (g/cc) | -0.074 | --- | **-0.743** | **---** | +0.333 | --- |
| Mechanic impedance (KPa) | +0.288 | --- | **-0.533** | **---** | -0.022 | --- |
| Slope (%) | --- | +0.281 | --- | +0.398 | **---** | **-0.748** |
| Undergrowth plant canopy (%) | -0.229 | -0.402 | +0.213 | +0.074 | **-0.592** | **-0.615** |
| Stoniness (%) | +0.162 | --- | -0.266 | --- | **-0.606** | **---** |
| Bare soil (%) | +0.164 | +0.507 | -0.102 | +0.141 | **+0.833** | **+0.684** |
| Absolute total variance (%) | 28.76 | 35.82 | 25.00 | 26.73 | 12.78 | 10.24 |
| Accumulative total variance (%) | 28.76 | 35.82 | 53.76 | 62.55 | 66.54 | 72.78 |

Three principal components were identified and variables grouped to FI, FII and FIII. Percentages of the explained variability and sense of correlation (+ or -) of each variable in each component and percentages of the explained total variability of each variable by the three components are presented.

The three components, described approximately 70% of the total variance of the observations, and each one was basically characterized by the same set of variables. A correlation factor higher than 0.5 was considered significant for the variables values characterizing each factor. The FI represented 31% and 39%, the FII 25% and 27% , and the FIII a 11% and 9% of the mean absolute total variance in summer and winter respectively. Concerning the studied rainfall events and seasons, no significant differences were found.

Table 5.13. Weight of each soil parameter in each constructed soil quality index

| **Index** | **Variables** | **Weight** | | | |
|---|---|---|---|---|---|
| | | **Summer** | | **Winter** | |
| | | **P max** | **P min** | **P max** | **P min** |
| **FI** | Runoff volume (l/m$^2$) | 0.110 | 0.080 | 0.105 | 0.091 |
| | Dissolved calcium (mg/m$^2$) | 0.103 | 0.097 | 0.102 | 0.091 |
| | Dissolved magnesium (mg/m$^2$) | 0.115 | 0.097 | 0.103 | 0.091 |
| | Dissolved potassium (mg/m$^2$) | 0.108 | 0.065 | 0.102 | 0.091 |
| | Dissolved sodium (mg/m$^2$) | 0.082 | 0.097 | 0.089 | 0.090 |
| | Dissolved organic carbon (mg/m$^2$) | 0.096 | 0.099 | 0.103 | 0.091 |
| | Dissolved nitrogen (mg/m$^2$) | 0.100 | 0.093 | 0.101 | 0.091 |
| | Eroded soil (g/m$^2$) | 0.107 | 0.099 | 0.104 | 0.091 |
| | Eroded organic carbon (g/m$^2$) | 0.107 | 0.104 | 0.098 | 0.091 |
| | Eroded nitrogen (g/m$^2$) | 0.072 | 0.100 | 0.093 | 0.091 |
| | Vegetal debris (g/m$^2$) | --- | 0.069 | --- | 0.091 |
| | | 1.000 | 1.000 | 1.000 | 1.000 |
| **FII** | Aerial plant canopy (%) | 0.105 | 0.152 | 0.134 | 0.126 |
| | Water holding capacity (%) | 0.130 | 0.143 | 0.166 | 0.168 |
| | Sand fraction (%) | 0.126 | 0.140 | 0.119 | 0.134 |
| | Clay fraction (%) | 0.080 | 0.120 | 0.092 | 0.085 |
| | Cation exchange capacity (cmol/Kg) | 0.112 | 0.148 | 0.159 | 0.160 |
| | Soil organic matter (%) | 0.139 | 0.148 | 0.163 | 0.163 |
| | Soil total nitrogen (%) | 0.120 | 0.149 | 0.167 | 0.164 |
| | Bulk density (g/cc) | 0.109 | --- | --- | --- |
| | Mechanic impedance (KPa) | 0.078 | --- | --- | --- |
| | | 1.000 | 1.000 | 1.000 | 1.000 |
| **FIII** | Slope (%) | --- | 0.365 | --- | 0.396 |
| | Undergrowth plant canopy (%) | 0.291 | 0.300 | 1.000 | 0.281 |
| | Stoniness (%) | 0.298 | --- | --- | --- |
| | Bare soil (%) | 0.410 | 0.334 | --- | 0.322 |
| | | 1.000 | 1.000 | 1.000 | 1.000 |

Pmax: maximum recorded rainfall in the study season; Pmin: minimum recorded rainfall in the study season. Each soil variable has been weighted on technical criteria.

The contribution of each significant variable to its correspondent factor are very similar, and thus all parameters can be considered equally important to the factor (Table 5.13.).

### 5.2.5. IDENTIFICATION OF THE MAIN FACTORS OF LAND DEGRADATION AT PLOT SCALE

The identification of the main factors of land degradation at plot scale has been based on the previous conducted assessments and in particular on the one related to the development of soil quality indices. This latter appraisal has involved the analysis of the overall dataset from all assessments, including physico-chemical variables, factors related to the vegetation cover and to soil erosion by water (section 5.2.4.). Therefore, it is suggested that the set of variables revealed as significant in the previous test, may be considered the main factors of soil quality or degradation at plot scale for the target area. Summarising these factors may be listed as follow: rainfall amount, runoff generation, dissolved calcium, magnesium, sodium and potassium, dissolved organic carbon and nitrogen, soil erosion, eroded organic carbon and nitrogen, vegetal debris, aerial plant canopy, water holding capacity, particle size distribution and therefore textural class, cation exchange capacity, soil organic matter, total nitrogen, bulk density, mechanic impedance, slope angle, undergrowth plant canopy, bare soil and stone cover. Conversely, the previous conducted assessment show that the following factors will not be considered important aspects of soil quality at plot scale for the specific study area: electrical conductivity, pH, base saturation, soil depth and soil moisture content.

## 5.3. DEVELOPMENT AND APPLICATION OF THE DIS MODEL

### 5.3.1. IDENTIFICATION OF THE MOST RELEVANT INDICATORS OF LAND DEGRADATION

The general identification of the most relevant indicators of land degradation and desertification has been undertaken in chapter three.

## 5.3.2. DEVELOPMENT OF A SET OF POTENTIAL INDICATORS OF DESERTIFICATION

According to Stocking and Murnaghan (2001), land-degradation indicators may be drawn from any aspect of how the quality of land degrades. Here, since soil degradation is considered the main process of land degradation as well as desertification, the study has been focus on indicators of soil condition (state indicators), which are mainly physical and chemical.

### 5.3.2.1. Potential indicators at the plot scale

The overall set of assessed variables (section 5.2) represents most of the required characteristics for being effective indicators. In general, parameters are easy to quantify through monitoring activities or special measuring, and therefore data are relatively easy to obtain. However, some of them are very time-consuming to measure (e.g. cation exchange capacity). Due to economic restrictions, determinations were conducted in the most cost-effective way, given the low budget, which was mainly invested in fieldwork equipment and campaigns, and laboratory analysis. Thus, these variables are considered cost-effective at plot scale. Moreover, although the methodology used for the determinations was simple, to a large extent results seem to be reliable, since they are logical and coherent in comparison with other studies of similar characteristics (e.g. Romero-Díaz et al., 1999; Martínez-Fernández et al., 1995). Here, the whole set of parameters may be considered understandable, that is to say, clear, simple and unambiguous, although some variables are better known as proper indicators than others (e.g. soil organic matter content). Regarding the relevance of the studied parameters, the development of the soil quality indices (section 5.2.4), has helped in the identification of the most important variables for soil degradation assessment by grouping them into three different factors (FI, FII and FIII). In this context, parameters such as: soil erosion, eroded organic carbon, eroded nitrogen, runoff generation, dissolved organic carbon, dissolved nitrogen and dissolved cations ($Ca^{2+}$, $Mg^{2+}$, $K^{+}$ and $Na^{+}$) are considered relevant for assessing land-degradation processes. Variables such as water holding capacity, sand and clay percentage, cation exchange capacity, soil organic matter content, soil total nitrogen and to some extent bulk density and mechanic impedance, account for soil quality characteristics. Finally, parameters such as slope angle, plant canopy, bare soil, are relevant when assessing the role of the vegetation cover as soil protection.

On the other hand, since the aforementioned variables were measured and addressed at the plot scale, their analyses provide evidence that they are significant at this particular scale. Here, the overall set of soil-water determinations was performed on the basis of a one-year established study period (the life time of the bounded runoff-erosion microplots), and therefore results from this assessment cannot support alone the consideration of these variables as appropriate land degradation indicators, since according to Norton et al. (1999), the analysis should have been conducted over a larger period of time, between one and three years. However, the assessment has demonstrated its validity for the identification of potential effective indicators of land degradation, since there is a clear pattern in most of the variables which results when analysing differences among seasons, land uses, environments and rainfall events.

#### 5.3.2.2. Potential indicators at the catchment to regional scale

As previously mentioned, a potential effective indicator, in addition to be quantifiable, understandable, reliable, cost-effective and relevant, must also be either measurable or able to be reliably modelled at a policy-relevant scale, in order to be useful for the specific requirements of local communities and decision-makers, who have responsibility for areas above the field scale.

Therefore, the identified potential variables at the plot scale likely to be considered appropriate indicators of land degradation are significantly reduced. On the one hand, most of the parameters are neither easy to measure nor to monitor at a policy-relevant scale (such as the catchment scale). Hence, data not readily available would be obtainable at a cost to benefit ratio which is too high due to constraints of time and logistics, personnel and technical issues. On the other hand, only a few group of variables are easy and viable to model, considering the required input variables and the complexity of the mathematical equations involved.

Thus, the potential indicators at a policy-relevant scale selected on the basis of the plot scale hydrological and soil analysis are runoff generation and soil erosion, which are processes of land degradation that are relatively easy to measure and model at this scale and another group of variables which include slope angle, slope aspect, bare soil, vegetation cover, particle-size distribution, and bulk density which can be assessed via remote sensing. All of these parameters will be used in the development of the Desertification Indicator System (DIS) for the Serra de Rodes catchment, which will involve geographical information systems (GIS) and modelling techniques.

### 5.3.3. DEVELOPMENT OF SPATIAL DATASETS

The advantages of using geographical information systems (GIS) in the development of a Desertification Indicator System (DIS), relevant at a policy scale are broadly described in chapter three.

#### 5.3.3.1. Digital Elevation Model (DEM), Slope, Aspect and Land use maps

The Digital Elevation Model (DEM) was developed on the basis of the topographic maps of the study area. The contour lines were manually digitised at an interval of 5 metres, leading to possible errors in terms of precision and accuracy in the output map. The 5 metres contour line interval determined the spatial resolution of the DEM entailing a cell size of 5 metres, as a default in all the maps used in the development of the DIS model. The general characteristics of each of the created maps are:

| | | |
|---|---|---|
| **Cell size:** | 5 metres | |
| **Rows:** | 397 | |
| **Columns:** | 361 | |
| **Projection:** | Universal Transversa Mercator (UTM) | |
| **Coordinates:** | | |
| | Left corner: | 508687.279 |
| | Right corner: | 510492.279 |
| | Bottom: | 4686509.3308 |
| | Top: | 4688494.3308 |

Once the DEM has been created, general information may be easily calculated from it through the use of GIS. Some of the available information is: the slope, aspect, hill-shading, flow direction and basin. Here, the information derived from the DEM (Figure 5.5) and needed in the DIS model is the slope map (Figure 5.6), which displays the grade of steepness expressed in degrees and the aspect map (Figure 5.7), which identifies the down-slope direction. Slope controls flowpaths and the magnitude of erosion, aspect controls the radiation receipt and thus evaporation.

The Land use map (LUM) was created on the basis of aerial photographs of the study catchment (1:5000, 1996) by visually identifying the main land uses and then manually digitising and labelling them accordingly.

Hence, since the boundaries of each land use were not always completely clearly to recognizable in the aerial photographs due to their print quality, sometimes the identification was difficult leading to possible small errors. However, in general, the digitising of the principal land use polygons can be considered reliable, accurate and precise. Several maps needed in the several equations composing the DIS such as: the saturated hydraulic conductivity, potential infiltration capacity, vegetation cover percentage, soil depth, and soil porosity, were created on the basis of the LUM.

The next step in the development of the DIS will be the identification of the processes, which will constitute the hydrological and soil-erosion model, and consequent selection of the most appropriate mathematical equations for describing each of these processes.

Accounting for the importance and extension of this procedure, chapter six is completely focused on the description of the DIS model. The sensitivity analysis of the DIS comprises chapter seven and the validation as well as the description of the land use scenarios are exposed in chapter eight.

**Digital Elevation Model (DEM)**

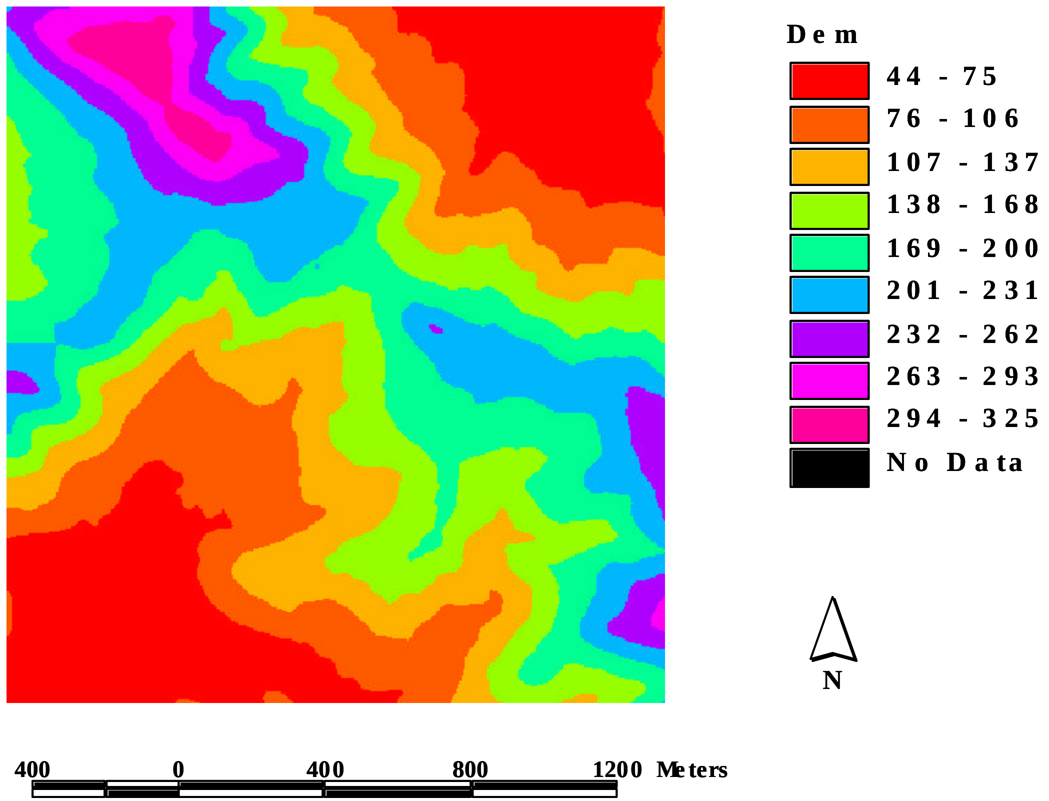


Figure 5.5. Digital Elevation Model (DEM) of the overall study catchment

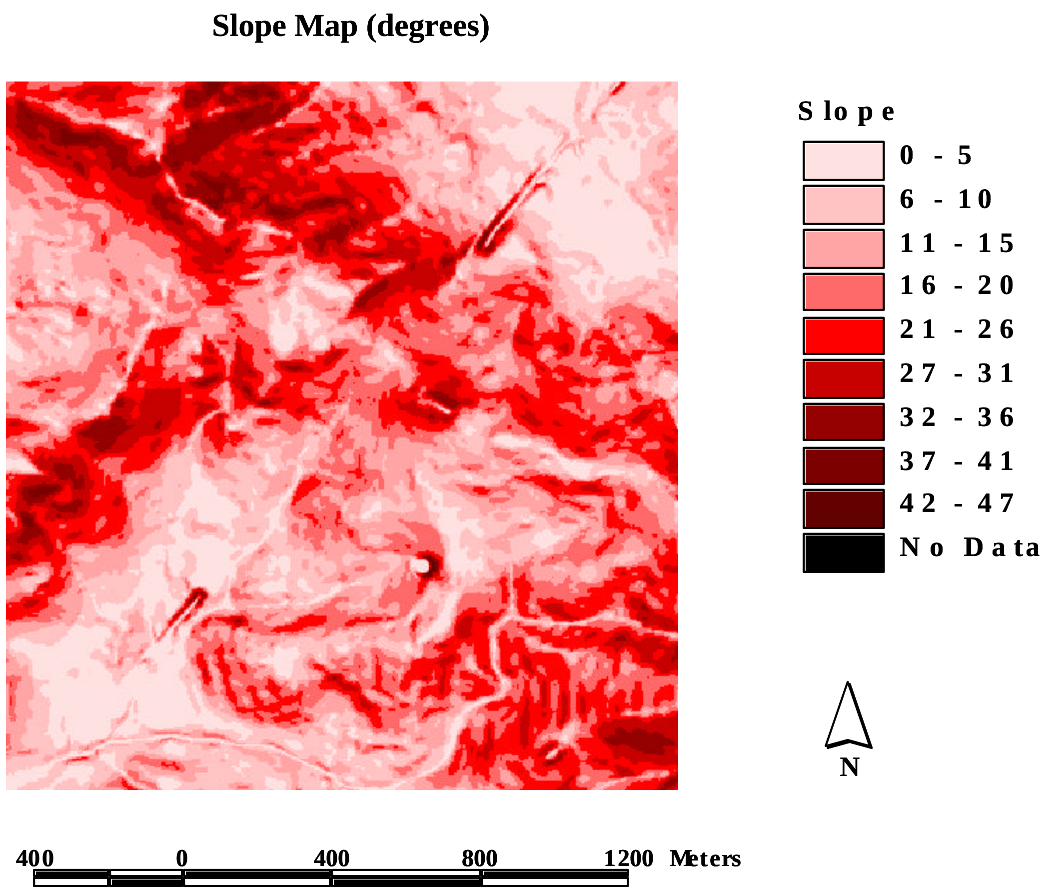


Figure 5.6. Slope map of the Serra de Rodes catchment

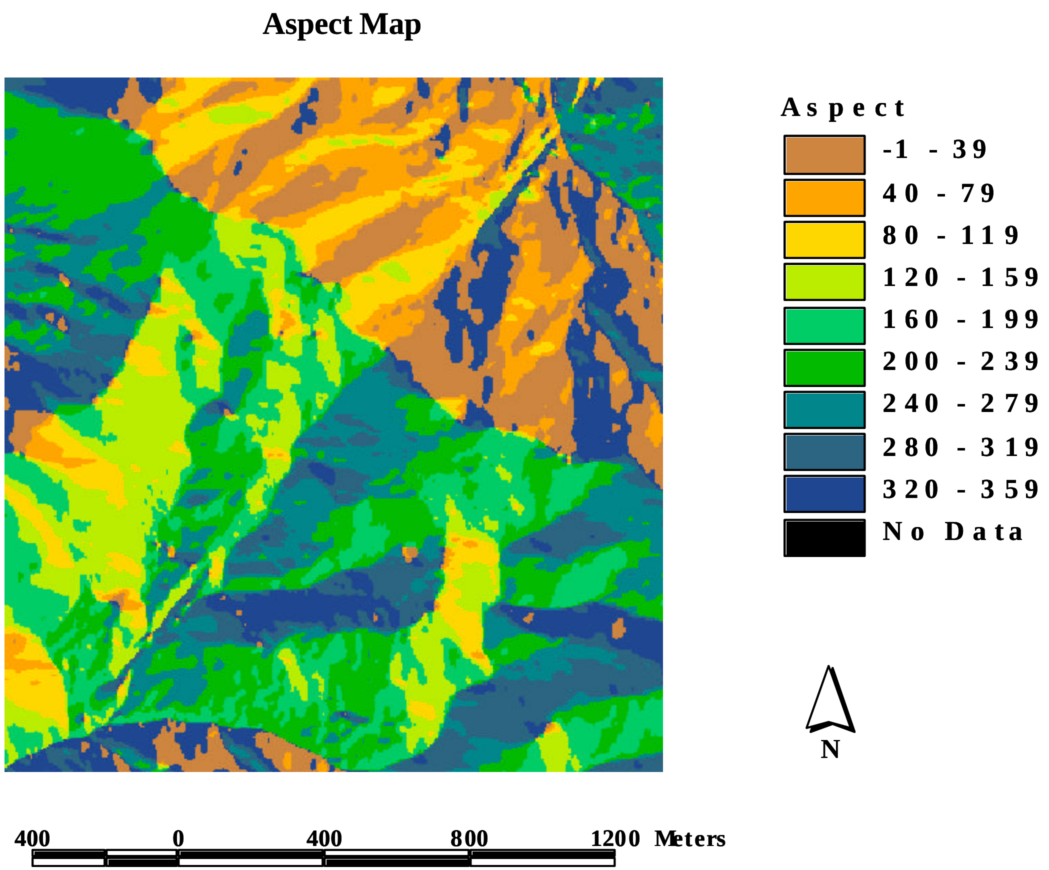


Figure 5.7. Aspect map of the Serra de Rodes catchment

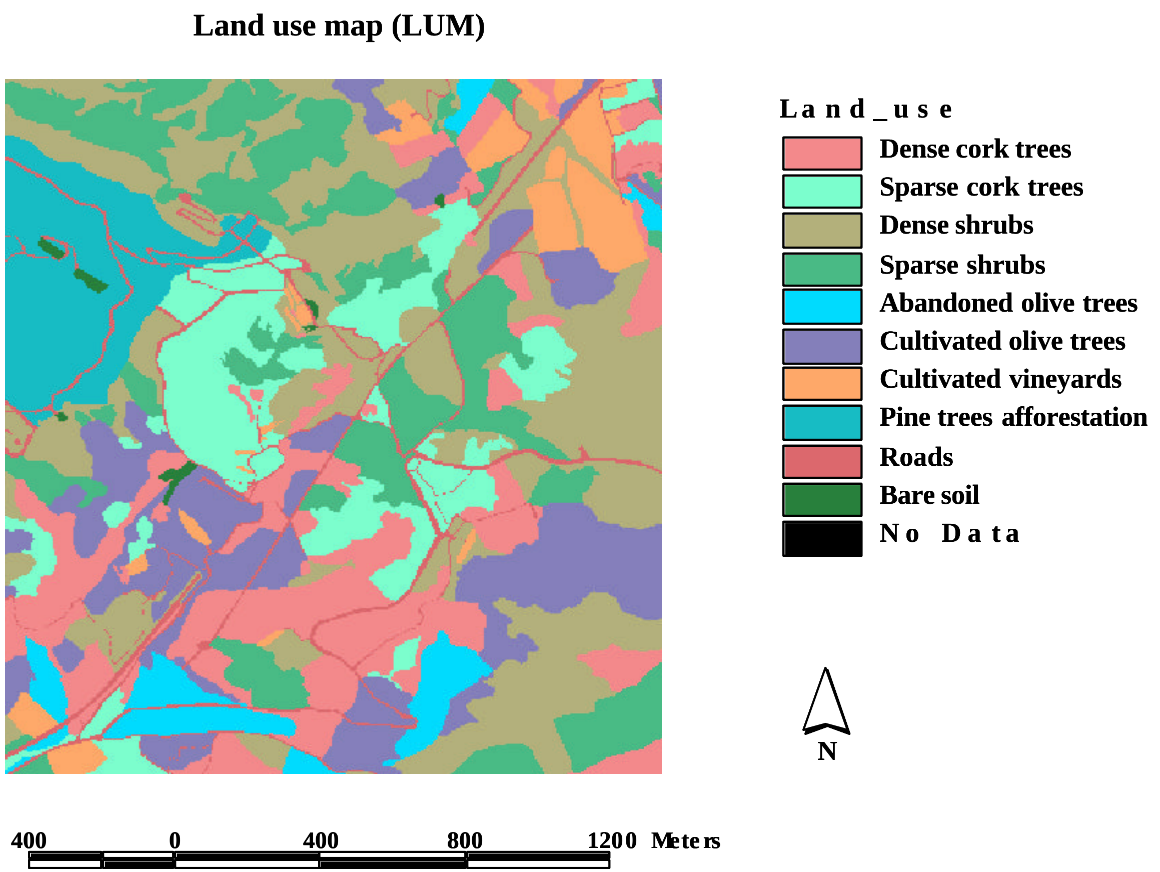


Figure 5.8. Land use map of the study area.

# 6. DESCRIPTION OF THE DESERTIFICATION INDICATOR SYSTEM (DIS)



## 6.1. WHY MODEL?

### 6.1.1. INTRODUCTION

Modelling is a tool of increasing importance in environmental research because powerful computers make the development of new techniques (e.g. GIS) possible. In general, models simulate the effect of an actual or hypothetical set of processes, and forecast or predict one or more possible outcomes, although they can never represent exactly the real world (Kirkby et al., 1993). Thus, models are used to describe, explore and analyse how a system works, aiming at representing this system of interest as accurately and precisely as possible (Hardisty et al., 1993). In this same context, the creation of a model always entails decisions and compromises because on the one hand that the model has to be a simplified representation of the complex real system of study, and on the other hand, all the factors considered not relevant according to the objectives of the project must be omitted (Cullen and Frey, 1999).

In particular, environmental models can be more or less complicated according to the requirements of the input data. Although these models might provide outputs not directly relevant to a specific risk management objective, they can be helpful in improving essential insights regarding risk processes and in learning lessons useful for further model development (Frey and Patil, 2001). Here, the model concerns desertification and land-degradation processes in a Mediterranean environment and is developed with the aim of, not only improving the understanding of the structure and function of this complex system, but also making a contribution towards decision-making concerning it.

### 6.1.2. ENVIRONMENTAL MODELLING

The necessity of using models in environmental research arises from the fact that much detailed knowledge and data are collected in environmental field research but in most cases it is not known what their consequences are for the whole system. This is especially the case because practical field measurements are usually only possible at the plot scale whilst providing information for environmental decision-making is usually a catchment or regional scale process. Models can combine these detailed information and data with knowledge of processes in a logical manner and integrating behaviour for the whole system from them (Jakeman et al., 1993). Environmental field data are often collected at some representative locations and over a short period of time. In this context, models can use these local and short-term data and can be used to deduce environmental dynamics for larger areas and over longer periods of time. One of the objectives of modelling is to help to get a better understanding of the structures and mechanisms operating in environmental systems (Hardisty et al., 1993). Moreover, models can be used to develop new hypotheses and reveal gaps in knowledge and consequently provide new input for environmental research.

There is often a considerable uncertainty in parameterisation data and some model processes (e.g. rainfall in semi-arid zones). However, if models have provided a good understanding of the environmental problem of interest they can be used as a very powerful tool for practical management problems. By the systematic investigation of the consequences of different management measures in the model one can determine the best management option. That means that investigations of models can be used as a substitute for experiments which cannot be performed in reality. Thus models can be used as robust decision-support tools for practical management problems.

Spatio-temporally variable systems of all scales can be handled by the powerful method of grid-based modelling. This can be combined with Geographical Information Systems (GIS) to model at the landscape scale (Fotheringham and Wegener, 1999).

## 6.2. THE MODELLING PROCESS: A GENERAL APPROACH

According to Dingman (2002b), Hardisty et al. (1993) and Kirkby et al. (1993), the general procedure (Figure 6.1.) for environmental modelling, can be summarised as follows:

**(a) Establishment of the model purpose:**

To clearly define what are the specific objectives to be accomplished by the development and running of the model.

**(b) Evaluation of the data availability against the data needs of the model:**

The nature and form of the available information about the system being modelled and the nature and form of the available input data will dictate the conceptualisation of the model. Ideally, data should be collected in the framework of hypothesis testing within which modelling can play a vital part. The model may provide the basis for testing, so that data collected under such circumstances would be appropriate for the model. Some data may be incorporated as input parameters and some other data will be used in the validation process of the model.

**(c) Conceptualisation of the problem:**

The first step in the creation of a model is to define a procedure, which must be expressible in logical or numerical form, for forecasting on the basis of your understanding of the environmental problem. Hence, the overall form and essential components of the model have to be determined on the basis of the scientific purpose of the model, and bearing in mind that this idea must be translated into an explicit formulation of the nature and form of the model output that is required. In this context, the type of information required has to be taken into account, as well as the required accuracy and precision of the output, the location for which the output is required and the time intervals for which the output is required.

**(d) Selection or development of the appropriate model:**

The choice of model type will depend on the aims of the project, the nature of the system to be modelled and the level of understanding of the processes operating in the system. Models can only be as good as the knowledge or understanding at the time of their construction (Thornes, 1989). The availability of appropriate data may be a major constraint (Hardisty et al., 1993). In modelling environmental systems the most appropriate type of model is usually a mathematical simulation model, defined as an explicit sequential set of equations and numerical and logical steps that converts numerical inputs representing flow rates or states of storages to numerical outputs representing other flow rates or storage states (Dingman, 2002b).

**(e) Selection of the appropriate computer language and/ or software:**

To a large extent, the selection of the appropriate computer language is subject to the type of model to be created. There exists a wide range of programming tools, from easy (e.g. Visual Basic) to more complicated ones (e.g. FORTRAN), in terms of management.

**(f) Representation of the processes:**

This part of the process aims at finding the best mathematical expressions for representing each of the selected processes or variables (components), which compose the overall model. Thus, parameterisation is a way of expressing an assumed relationship between variables in mathematical form. Estimates of some parameters may be based on observations or laboratory experiments (empirically derived) or based on scientific principles (theoretically derived). The selection of these equations has to take into account the equilibrium between the operational simplicity and the realistic representation of the system, as well as the availability, accuracy and the precision of the data.

**(g) Selection of the initial parameter values:**

Once the several components of the model are expressed in a mathematical form, the required input parameters are required. Some of these parameters may be physical constants and others are derived from fieldwork determinations or laboratory analyses.

**(h) Model debugging and verification:**

Once the model is written, it has to be run in order to test whether it is operational or not. The program must run without producing error messages. Furthermore the outputs of the model must be viewed in order to verify that they fall within reasonable limits for the variables being modelled (verification).

**(i) Acceptance testing or validation:**

The ultimate test of a model is validation, which is based on the comparison of the model with the real world system. Model validation is the comparison of model outputs with measurements of those same outputs taken under conditions identical to those used to parameterise the model. Where model outputs has been calibrated to a measured dataset, the validation data should be different to data used in calibration. The accuracy of the model is given as the difference between the measurement and model result for a particular variable.

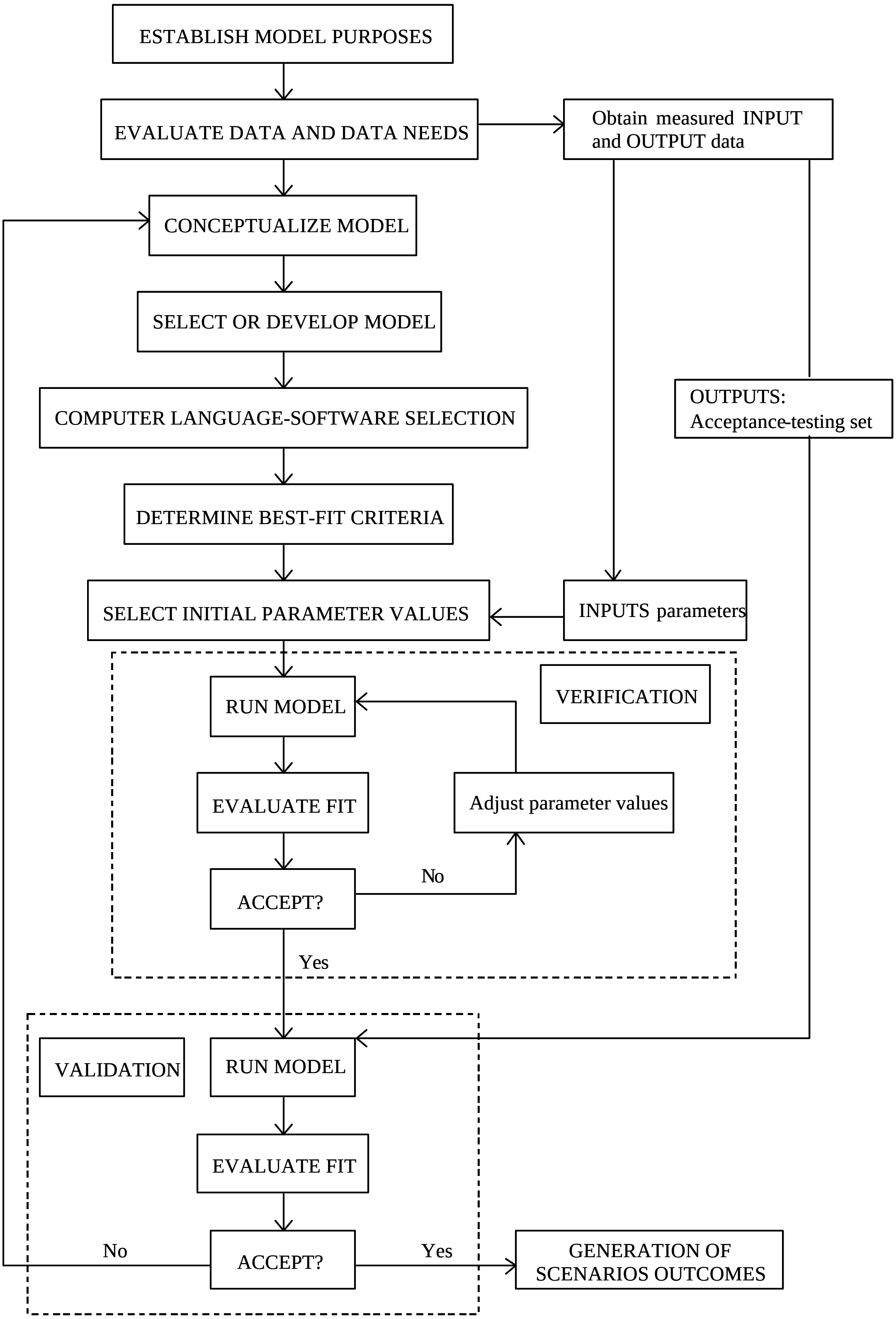


Figure 6.1. Flow chart for the modelling process (based on Kirkby et al., 1993).

## 6.3. THE DESERTIFICATION INDICATOR SYSTEM (DIS)

### 6.3.1. DIS MODEL PURPOSES

The purpose of developing the Desertification Indicator System (DIS) model is to help identify present and future threats for the development of effective measures to combat desertification effectively in the Serra de Rodes catchment. The model should be an information system able to monitor the main physical and chemical factors, as well as to assess the extent, intensity and severity of land degradation processes in this target area currently and for a series of policy scenarios. Therefore, the DIS will focus on the better understanding of the driving factors, which lead to desertification in the chosen area, and the factors likely to exacerbate the issue through the unforeseen implications of local scale policy decisions.

### 6.3.2. DATA EVALUATION AND DATA NEEDS FOR THE DIS MODEL

The data required for the development of the model can be divided into two types in accordance to their role in the overall modelling process: (a) input data; and (b) validation data. Regarding the input data, we distinguish physical and chemical data obtained from fieldwork determinations and/or laboratory analyses, meteorological data from an Automatic Weather Station (AWS), and finally, spatial data needed for the creation of geographical information, such as the Digital Elevation Model (DEM) and land use map (LUM). With reference to the model validation, mainly physical parameters measured during fieldwork campaigns were required. The overall set of data needed for the DIS model is listed in Table 6.1.

The accuracy and precision of several field measurements such as vegetation cover, runoff generation and sediment yield (from microplots), were subject to funding constraints, personnel availability and time constraints.

Table 6.1. Summary of the data needed for the DIS development

| | | |
|---|---|---|
| **INPUT DATA** | **Fieldwork-laboratory** | Soil depth; soil porosity; soil moisture; potential infiltration capacity; textural class; particle size distribution; bulk density; vegetation cover percentage |
| | **Meteorological** | Rainfall amount; mean air temperature; solar radiation at the land surface |
| | **Spatial data** | Topographic maps and aerial photographs |

Moreover, also due to funding restrictions, the equipment for recording meteorological data such as evapotranspiration, rainfall amount and intensity, and solar radiation, from the exact location of the study area, was not available and therefore, these data were obtained from an AWS located in sites nearby. The monthly temporal availability of the meteorological data dictated the monthly time scale of the overall required dataset as well as the monthly timestep of the model. Here, as for most of projects, the availability or unavailability of appropriate data was a major constraint in the model development.

### 6.3.3. CONCEPTUALISATION OF THE DESERTIFICATION ISSUE

Bearing in mind the conclusions of the literature review which provide knowledge of the environmental problem, the scientific purpose of the model, the nature of the required information, and the data availability; the overall structure and essential components of the DIS model were determined as in Figure 6.2.

Several sub-models were chosen in order to study runoff generation and soil erosion (outputs), deemed to be good indicators of the main land-degradation processes in the target area. Hence, the selected components are: (a) soil moisture; (b) potential infiltration capacity; (c) runoff; (d) actual infiltration capacity; (e) overland flow (routing of the runoff); (f) potential evapotranspiration; (g) actual evapotranspiration; (h) groundwater recharge; and (i) soil erosion. Each sub-model (process) was expressible in the form of an equation. Figure 6.2 shows the framework in which the modelled processes can be represented.

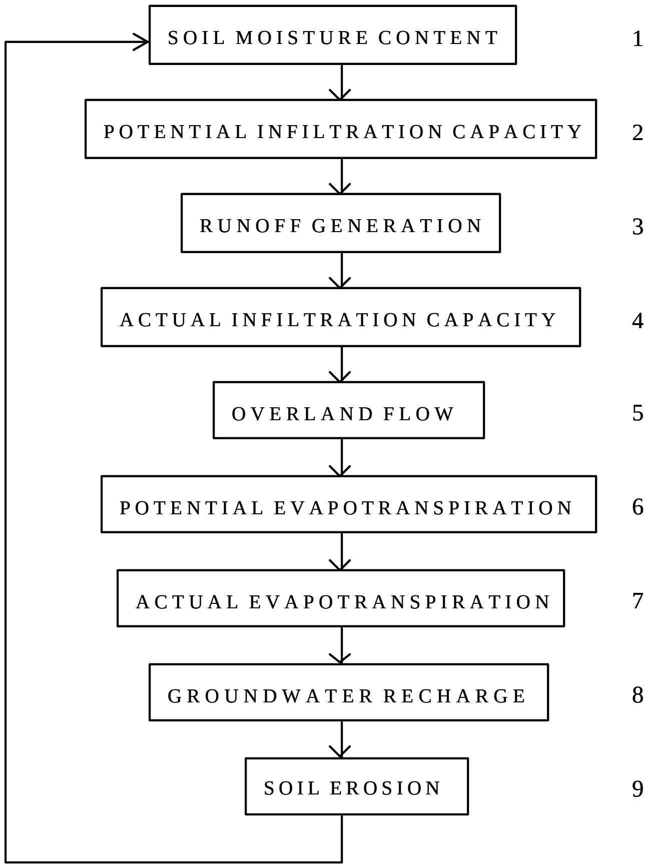


Figure 6.2. Flow diagram of the sequence of the several components of the DIS model

### 6.3.4. SELECTION OF THE APPROPRIATE MODEL TYPE

There are several classifications of model types. According to Kirkby et al. (1993), a model can be either *Deterministic* or *Stochastic*. A deterministic model is defined as a model that, for a given set of inputs, has a unique forecast. On the other hand, a stochastic model contains at least some random or chance element in the process operation or in the inputs to the model, so that more than one, and usually a very large number of outcomes are possible for a given input.

Regarding this first classification, the DIS model should be deterministic, since the several components addressing environmental processes are known and described and therefore non-random elements are present or included in the model. Furthermore, the

DIS is also defined as a *computer or mathematical model*, since all the sub-models are expressible in a numerical form and need a logical procedure for being implemented on a computer. According to Kirkby et al. (1993), deterministic computer models may be classified into three groups:

**(a) Black box models or Input/Output models:**

These are the simplest models, and they are usually defined experimentally or empirically. The internal workings of these models are not intended to directly represent the processes operating in the real world, even at an abstract mathematical level.

**(b) Process models:**

These models describe the mechanisms of particular operations which occur in the real world by not only linking the input and output from a process but also by including the mechanisms underlying these processes and how they might be expressed mathematically. In this context, the understanding of the processes helps to direct fieldwork in the choice of variables. They are built up from a flow diagram, which represents the physical storages and/or flows of energy or material in the real world. They can never be perfect or complete, but they are an attempt to make the conceptual behaviour of the model resemble the real world more closely than in a black box model. In most of these models, there are many processes operating at once and interacting with each other. As a result, the model is considered the sum of the overall set of sub-models, where each sub-model represents a single process or group of processes. These sub-models need to deal with flows of the same commodity, usually mass or energy; and deal with similar levels of space and time resolution.

**(c) Mass balance models:**

Bearing in mind that mass and energy are neither created nor destroyed, this kind of model are constrained by some form of the “storage equation”: Input-Output = net increase in storage. Mass balance models provide the framework for a physically based model, within which the individual process models are supported.

The selection of the most suitable type of model is determined by the aims of the project, but also the nature of the system to be modelled and the level of understanding of the processes operating in the system. So, among the three types of deterministic computer models, the process-based model was found to be the most appropriate one for this thesis. Furthermore, the DIS model should be easy to monitor, manage and understand, and particularly convenient for policy decision-making purposes at the catchment-landscape scale. In this context, a spatial component was incorporated to the model, through the use of GIS, considered a powerful tool for managing resources, understanding and predicting complex and changing systems. Therefore, the resultant model was numerical, spatial and process-based.

### 6.3.5. COMPUTER SOFTWARE SELECTION

The development of the DIS required both GIS and modelling techniques, in order to accomplish the aims of the project. The first task was the creation of the DEM and the LUM of the study area, which were developed using ArcView 3.2., and mainly Arc/Info 8.0.2. GIS software packages (see chapter four). Both resultant maps are presented in chapter five (section 5.3.3.).

The kind of DIS envisaged here required the combination of GIS and modelling tools needed for its development. The *Arc Macro Language (AML)*, a computer language that runs in the Arc/Info environment, was chosen at an early stage because of the following advantages:

1. AML is a native and robust programming language of the Arc/Info Workstation GIS software.
2. AML provides the same general functions as other programming languages.
3. AML provides a wide variety of operations to complement ARC/INFO. Almost anything done in ARC/INFO can be aided by the use of AML, and some tasks can be completely handled in AML.

4. AML allows the access to a range of functionality, from automating common tasks in Arc/Info, to creating complete graphical user interface-based, multithreaded applications.
5. AML is interpreted at run-time rather than compiled, making AML somewhat slow, but also easy to debug and modify. It also does not depend on platform-specific executables.

However, AML can be confusing and difficult to learn, but the main problem for using this programming language on the overall development of the DIS model, came at a later stage, when dealing with the routing operation for transport of the runoff water and eroded sediment. Since AML is not specifically designed for hydrological purposes, this operation was impossible to perform. This problem was overcome by the selection of a more suitable programming language, the *PCRaster Environmental Software (version 2.0)*.

Essentially, PCRaster is not only a GIS, consisting of a set of computer tools for storing, manipulating, analysing and retrieving geographic information, but it is also provided by an hydrological modelling component. The main reasons for its selection as the software for developing the DIS model are:

1. It permits the integration of environmental modelling functions with classical GIS functions such as database maintenance, screen display and hard copy output.
2. It is a raster-based system that uses a strict data-type-checking mechanism.
3. GIS and modelling functions are incorporated in a single GIS and modelling language for performing both GIS and modelling operations.
4. The exchange of ASCII files with any other modelling or GIS packages can be done in PCRaster and it is easy to perform through conversion operations (e.g. data conversion from PCRaster to Arc/Info and/or ArcView and vice versa).

5. Spatial modelling tools which allow dynamic process modelling over space and time through a scripting generic modelling language.
6. It can be used for building a wide range of models, from very simple up to conceptually or physically based models for environmental modelling.
7. It has geomorphological and hydrological functions (e.g. routing operations)
8. The results can be obtained in spatial and numerical format, making the data treatment easier.
9. The software is low cost and includes a complete help manual for an easy understanding and effective management.

Therefore, the PCRaster software provided the required tools for the development of the overall tasks involved in the DIS model, by incorporating a modelling component to a GIS, and allowing the generation of potential scenario outcomes and evaluations of the alternatives.

#### 5.3.5.1. The PCRaster script

The procedure for writing a PCRaster script consists of separate sections, where each section contains a certain functional part of the script. The basic sections for building a sequential model are: (a) the *binding section*, which binds the database file names to the names of each variable in the model; (b) the *areamap section*, which defines the model area and modelling resolution through a clone map which specifies the geographical location attributes of the maps used throughout the program. All the maps used or generated by the model have the location attributes of the clone map; (c) the *timer section*, which controls the time dimension of the model, specifying the duration of a model run by setting the time at the start and end of a model run; (d) the *initial section*, corresponds to the static or cartographic modelling script. Here, the spatial (maps) or non spatial (constants) attribute values at the start of the model run are given; and (e) the *dynamic section*, defines the operations performed for each timestep of the model run.

### 6.3.6. BEST-FIT CRITERIA DETERMINATION

The criteria for selecting the best equation of each sub-model, were simplicity and reliability of the model. This selection was subject to several factors such as data availability, precision and accuracy, as well as the input data required for the numerical equations. As previously mentioned in chapter four, the temporal scale of the available meteorological data determined the timestep (t) of the model, set up as monthly. For this same reason, all the mathematical equations had also to be chosen accordingly, and are therefore, monthly. On the other hand, the selection of PCRaster, as the best software for the creation of the model placed some constraints on the model that could be built, particularly because PCRaster does not allow nested iteration or loops.

#### 6.3.6.1. Soil-moisture content sub-model

Soil moisture content was the first sub-model calculated in the DIS model (Figure 6.2). According to the detailed selection criteria, the best expression for representing this component was one based on soil-water balance.

$$\text{Moist}_{(t)} = \text{Moist}_{(t-1)} + ((\text{AINF}_{(t)} - \text{ActET}_{(t)} - \text{Recharge}_{(t)}) / (\text{Depth}_{(t)})) \qquad (6\text{-}1)$$

$0.001 > Moist_{(t)}\ ?\ Porosity$

$\text{Moist}_{(t)} = \text{Porosity}; \quad Moist_{(t)} > Porosity$

$\text{Moist}_{(t)} = 0.001; \quad Moist_{(t)} = 0.001$

$\text{Moist}_{(t)}$ = soil moisture content at timestep t (mm water/mm soil)

$\text{AINF}_{(t)}$ = actual infiltration capacity of the soil at timestep t (mm water/month)

$\text{ActET}_{(t)}$ = actual evapotranspiration at timestep t (mm water/month)

$\text{Recharge}_{(t)}$ = groundwater recharge at timestep t (mm water/month)

$\text{Depth}_{(t)}$ = soil depth at timestep t (mm soil)

(t) = current timestep of the model (month)

The depth of the soil in the equation was needed for converting the units from mm water/month to mm water/mm soil (volumetric). The first restriction of the above equation is based on the assumption that in the real world, soil moisture cannot exceed soil water storage capacity. Moreover, a second restriction was required in the soil moisture sub-model for the correct performance of the overall model. Soil moisture is involved in the calculation of several other sub-models and vice versa. To avoid overflow errors caused by division by zero in equations which divide by soil moisture (and to maintain logical consistency in the model) the soil moisture must not be allowed to become zero or negative respectively but must be held at a minimum which is slightly positive or above. Similarly inputs of soil moisture cannot exceed the capacity of the soil to store water thus soil moisture is capped at the value of soil porosity. Therefore, the last restriction accounted for the excess of water when soil moisture is greater than soil porosity.

$$\text{RFMoist}_{(t)} = (\text{Moist}_{(t)} - \text{Porosity}) * \text{Depth}_{(t)}; \quad \textit{Moist}_{(t)} > \textit{Porosity} \qquad (6\text{-}2)$$
$$\text{RFMoist}_{(t)} = 0; \quad \textit{Moist}_{(t)} = \textit{Porosity}$$

$\text{RFMoist}_{(t)}$ = water surplus at timestep t (mm water/month)
Porosity = soil porosity (mm porespace/mm soil)

This water surplus represents the runoff generation (its value is added to the runoff total). Here soil depth (mm soil) was used for converting soil moisture and porosity units (mm water/mm soil and mm porespace/mm soil, respectively), to mm water/month. Thus, the RFMoist will be in the same units as the runoff water.

#### 6.3.6.2. Potential infiltration capacity sub-model

Infiltration is a complex process mainly controlled by soil properties and the supply of moisture at the surface (Maidment, 1993). The *Green-Ampt model* (Swartzendruber, 1974; Salvucci and Entekhabi, 1994), a simple, theoretical approach to the infiltration process, was initially selected in the development of the DIS. However, a good performance of this physically based model, required as input data a value of rainfall intensity too high for the Mediterranean characteristics, bearing in mind that in the study area the mean annual rainfall is approximately 600 mm. Therefore, the model was considered not to perform well in terms of realistic results.

The second tested model for the infiltration process was the *HYDROPAR spreadsheet model* (Mulligan, 1996). The equations of the parameters used to calculate infiltration in the HYDROPAR model were based on the work presented in Campbell (1985), which is physically based and has been tested on large data sets. The infiltration process was individually calculated for each of the selected land uses, as a relationship between soil-moisture content and hydraulic conductivity of these soils. In all the cases, a polynomial equation described this relationship, as well as the potential infiltration capacity of the soil. This model had the advantage of using equations which are simple and easy to parameterise yet which are physically based, and therefore, it accounted for the soil characteristics that most influence the infiltration process (e.g. saturated hydraulic conductivity and bulk density)

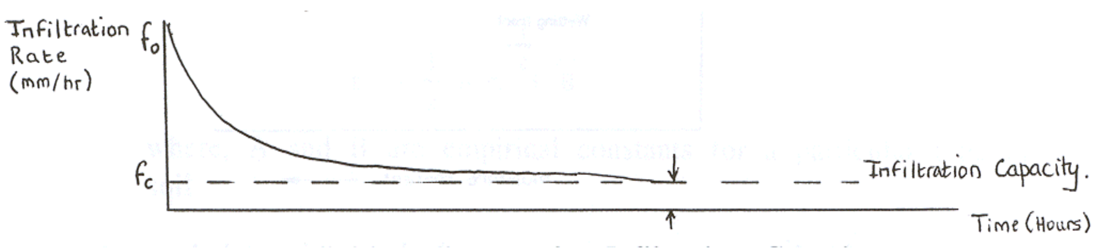


Figure 6.3. Horton's definition diagram for infiltration capacity

However, the estimated results were considered neither reliable nor precise or accurate, after comparing them with the available experimental infiltration data (fieldwork), since they were too low for the characteristics of the soils of the study area. Bearing in mind that the HYDROPAR model only accounts for the microporosity of the soils through input parameters such as soil textural class, particle size distribution and bulk density, the macroporosity in the study of these particular sandy-loam or loamy-sand soils, was found to have an essential role on the infiltration process. Therefore, this model was not appropriate for the estimation of the potential infiltration capacity.

The most effective model for describing and calculating infiltration was found to be the *Horton's equation* (Horton, 1940; Cerdà, 1996). This equation is based on the principle that when rain falls on a dry soil, the initial rate of infiltration is high because gravity and soil suction due to capilliarity both work together to draw water into the soil ($f_0$); and as rainfall continues and the soil becomes wetter the rate at which it takes up water decreases to a constant value, the infiltration capacity (fc) (Figure 6.3.). The infiltration capacity is mostly dependent on the type of soil, being high for sandy soils and low for clay soils. Specific controls on infiltration operate both at the soil and sub-surface. Surface controls are the vegetation cover (interception, humus level and soil fabric) and the soil surface condition (surface sealing, compaction, impermeable pans and cracks and fissures). In relation to sub-surface controls these are the soil profile, moisture gradient and cracks and macropores. Horton (1940) was the first to infiltration versus time. However, in the DIS model, soil infiltration is expressed as a function of the soil moisture content, which is the equation used in the HYDROPAR model (Mulligan, 1996):

$$PotInfil_{(t)} = SatHK + (Pot_{(t)} - SatHK)^{(-h * Moist\ (t))} \qquad (6\text{-}3)$$

$PotInfil_{(t)}$ = potential infiltration capacity of the soil at timestep t (mm water/hour)
SatHK = saturated hydraulic conductivity of the soil (mm water/hour)
$Pot_{(t)}$ = experimental soil potential infiltration capacity (fieldwork) per each selected land use at timestep t (mm water/hour)
h = empirical value from the relationship between soil moisture and potential infiltration capacity of the soils of the several land uses. It depends on the soil type (Hortonian constant) (dimensionless)
$Moist_{(t)}$ = soil moisture content at timestep t (mm water/ mm soil)

Later equations reject Horton's approach and used a variety of alternatives including diffusion equations and ones based on the physical processes involved in water flow through a porous medium. However, logical and reliable infiltration results were obtained using Horton's equations for the particular study.

#### 6.3.6.3. Runoff sub-model

Based on the fundamental assumption that runoff occurs when the rainfall intensity exceeds the infiltration rate of the soil (Horton, 1940), runoff was estimated as the difference between rainfall and infiltration, using a simple and basic equation.

$$\text{Runoff}_{(t)} = \text{Rainfall}_{(t)} - \text{Infiltration}_{(t)}; \quad \textit{Infiltration}_{(t)} < \textit{Rainfall}_{(t)} \tag{6-4}$$
$$\text{Runoff}_{(t)} = 0; \quad \textit{Infiltration}_{(t)} = \textit{Rainfall}_{(t)}$$

There exist two possible scenarios: (a) the generation of runoff, when rainfall is greater than the potential infiltration capacity of the soil, and (b) the non-existence of runoff, when rainfall is equal or lower in amount, than the soil potential infiltration capacity. Runoff is also generated when the soil is saturated through the mechanism defined above in equation (6-2) and also because a saturated soil has an infiltration of zero so any rainfall will be converted to runoff.

As mentioned, rain data was available monthly (mm water/month), whereas PotInfil was in mm water/hour. Thus, rain data was first converted from mm water/month to mm water/hour. This conversion was achieved through calculations with the rainfall volume and the RFD characteristic of the study area, both available from the Automatic Weather Station (AWS). The RFD was assumed to have a constant pattern over time, since it mainly depends on the regional climatic characteristics, which at the same time are generally considered invariable for a particular area. The RFD factor, determines each percentage of rainfall from the total monthly amount, which falls at each individual intensity recorded by the AWS. The procedure for estimating the runoff generation consisted of several steps:

1. Calculation of hortonian runoff volumes as the difference between the volumes of water falling at each intensity and the intensity of infiltration (the infiltration rate).

RmI(x)mmhr = ((x) – PotInfil); *RmI(x)mmhr > 0* (6-5)

RmI(x)mmhr = 0; *RmI(x)mmhr = 0*

RmI(x)mmhr = runoff water associated to the rainfall intensity (x) (mm water/ hour)

PotInfil = soil potential infiltration capacity (mm water/ hour)

(x) = rainfall intensity (mm water/ hour)

2. Calculation of the volume of water within a daily total that falls at each of the rainfall intensities represented in the RFD.

Run(x)mm = (Rain * (RFD /100)); *RmI(x)mmhr ? 0* (6-6)

Run(x)mm = 0; *RmI(x)mmhr = 0*

Run(x)mm = runoff water associated to the rainfall intensity (x) (mm water)

Rain = monthly rainfall amount (mm water/ month)

RFD = percentage of the total rainfall amount fallen at a intensity (x) (%)

(x) = rainfall intensity (mm water/ hour)

3. Computation of final monthly runoff water.

$$\text{Runoff}_{(t)} = \overset{n}{\underset{i=1}{?}} \text{Run}(x_i)\text{mm} \qquad (6\text{-}7)$$

Runoff = monthly runoff water at timestep t (mm water/month)

Run(x)mm = runoff water associated to the rainfall intensity (x) (mm water)

(x) = rainfall intensity (mm water/hour)

RFMoist, is added to the result of Runoff, accounting for the contribution of water from moisture to the total monthly runoff.

#### 6.3.6.4. Actual infiltration capacity sub-model

The actual infiltration capacity process is the fourth sub-model to be measured. As for the runoff, this component was also calculated on the basis of the basic equation (6-4). Therefore, once the runoff water is determined, the actual infiltration process is easier to compute, since its value is considered as the remaining fraction of the total rainfall amount that has not contributed to the runoff generation. The procedure for this estimation is essentially the same as the conducted for the runoff:

AINF(x)mm = (Rain * (RFD /100)); *RmI(x)mmhr <= 0* (6-8)
AINF(x)mm = 0; *RmI(x)mmhr > 0*

AINF(x)mm = actual infiltration capacity at a rainfall intensity (x) (mm water)
Rain = monthly rainfall amount (mm water/ month)
RFD = percentage of the total rainfall amount fallen at a intensity (x) (%)
(x) = rainfall intensity (mm water/ hour)
RmI(x)mmhr = runoff water associated to the rainfall intensity (x) (mm water/ hour)

The total monthly actual infiltration capacity is computed as the sum of the overall set of AIN(x)mm.

$$AINF_{(t)} = \sum_{i=1}^{n} AINF(x_i)mm \qquad (6\text{-}9)$$

$AINF_{(t)}$= monthly actual infiltration capacity of the soil at timestep t (mm water/month)
AIN(x)mm = actual infiltration capacity at the rainfall intensity (x) (mm water)

(x) = rainfall intensity (mm water/hour)

#### 6.3.6.5. Overland flow sub-model

The aim of this sub-model is to calculate runoff on a cell-by-cell basis, through the local drainage direction map of the whole study area. In PCRaster, the estimation of this process is accomplished through the *accuthresholdflux* operator. Water is accumulated in the drainage network with transport limited by a threshold. Material less than the threshold is stored. This is the case for overland flow, which will only develop once a certain loss has occurred, saturating the soil. For each cell, the amount of material input, for instance the amount of rain, is given by runoff. This is transported in downstream direction through the consecutively neighbouring downstream cells, following the local drain directions. Each time material moves through a cell a certain amount is stored in the cell. The remaining material is transported out of the cell, these amounts of outflow from each cell into its neighbouring downstream cell are the result of the *accuthresholdflux* operator. This function moves all water to the catchment outlet in each timestep and takes no account of flow velocities. It is therefore most appropriate to situations where the timestep is sufficiently long to ensure that in the real world water will have had time to pass through the whole catchment. For shorter timesteps or very big catchments a kinematic wave approach is more appropriate. The overall procedure for the overland flow calculation is detailed in the appendix 1.

After the calculation of this process in the first timestep of the model, and due to the mathematical interrelationship of the several components of the DIS, the forthcoming sub-models (and their outcomes) will also be calculated on a cell-basis, instead of a land-use basis.

#### 6.3.6.6. Potential evapotranspiration sub-model

Evapotranspiration is a major component in terrestrial water balance and is often difficult to measure and predict. This process is a function of microclimate (weather conditions), soil water status and plant water use requirements. However, the calculation of the potential evapotranspiration (PET) only accounts for the atmospheric conditions, ignoring the plant and soil-water status and therefore, it does not take the limitation of soil moisture into consideration.

There are many different methods for estimating PET (Jacobs and Satti, 2001). The Penman-Monteith equation is considered as a standard method, although it requires the most extensive set of weather measurements and is also on of the most complicated to compute, since it uses complex unit conversions and lengthy calculations. This combination method consists of a radiation and an aerodynamic term, and is known as relatively accurate in both arid and humid conditions (Allen et al., 1998). However, in some places due to the lack of data (vapour pressure deficit (VPD), wind speed or solar radiation), PET cannot be calculated with the Penman-Monteith method, and empirical methods such as Hargreaves and Samani (1985), Priestley and Taylor (1972), or Turc (1961) should be considered as more reliable than Penman-Monteith calculated with estimations of VPD or any other variable (Allen et al., 1998).

The Penman-Monteith model was not applied here, since the required climatic data were not available for the study area. The selection of the best equation, not only accounted for the monthly temporal scale of the DIS model, but also for the complexity of the models in terms of quantity of parameters required. Then, two monthly-based models (Radiation methods), simpler to apply than the Penman-Monteith equation, were considered in this study, the Hargreaves (Hargreaves and Samani, 1985), and the Priestley and Taylor method (Priestley and Taylor, 1972). These radiation methods, provide a much more consistent estimate of PET than the temperature methods (e.g. Thornthwaite method), but they are less accurate and reliable than the combination methods (e.g. Penman-Monteith model). Radiation methods use a measure of solar radiation coupled with air temperature to predict PET, which here is considered controlled by available energy and the ability of evaporated water to be transferred from the surface. Both, the Hargreaves and the Priestley-Taylor method, are monthly based.

The relatively simple Priestley-Taylor method was selected and applied for PET estimation. Although, it is more complex than the Hargreaves method, the required climatic data were available or easily obtainable and it has the advantage of being a very commonly applied methods after the Penman-Monteith model.

The Priestley-Taylor equation (P-T) for the calculation of the PET is a simpler, semi-empirical model derived from the physics-based Penman-Monteith model (Monteith, 1965), and has been successfully applied in many areas (e.g. Rana, 1998). The P-T method took the concept of equilibrium evaporation as the basis for an empirical equation giving evaporation from a wet surface under conditions of minimal advection. It assumes a saturated atmosphere and adds an empirical term, the P-T coefficient to account for the fact that the atmosphere does not generally attain saturation. The P-T model is considered useful for conditions where the Penman-Monteith method cannot be applied due to the lack of required weather data. The first step for estimating PET through the Priestley-Taylor method was to calculate the solar radiation at the land surface (RADSPT), procedure which is detailed in appendix 2. The slope of saturation vapour pressure curve was another parameter required in the Priestley-Taylor equation, and its calculation is described in appendix 3.

Finally, the PET estimation through the Priestley-Taylor method is performed through the following equation:

$$PET_{(t)} = PTcoeff * (RADNET_{(t)} / Lambda) * (Delta_{(t)} / (Delta_{(t)} + Psychro)) \quad (6\text{-}10)$$

$PET_{(t)}$ = potential evapotranspiration at timestep t (mm water/month)

PTcoeff = Priestley and Taylor constant (dimensionless)
RADNET = net solar radiation map at the land surface ($MJ/m^2$ month)
Lambda = latent heat of water vaporization at a 20 ºC temperature (constant) (MJ/kg)
Delta = slope of saturation vapour pressure curve (kPa/ºC)
Psychro = Psychrometric constant (kPa/ºC)

The Priestley-Taylor method is considered to perform well given its relatively simplicity, and it is easier to parameterise than the widely used Penman-Monteith.

#### 6.3.6.7. Actual evapotranspiration sub-model

The soil moisture function plays a key role in deriving actual evapotranspiration (AET) from potential evapotranspiration (Kolka and Wolf, 1998). Potential evapotranspiration assumes that the soil is at field capacity and will overpredict evapotranspiration under drier soil.

The actual evapotranspiration function was developed connecting the potential evapotranspiration with the soil-moisture content. In this context, the simplest way but still reliable of calculating AET is through the assumption of a linear relationship between PET and soil-moisture content.

$$AET_{(t)} = PET_{(t)} * Moist_{(t)} \tag{6-11}$$

$AET_{(t)}$ = actual evapotranspiration at timestep t (mm water/month)
$PET_{(t)}$ = potential evapotranspiration at timestep t (mm water/month)
$Moist_{(t)}$ = soil-moisture content at timestep t (mm water/mm soil)

### 6.3.6.8. Groundwater recharge sub-model

For simplifying the calculations, it was assumed that the recharge to ground water in the study area is mainly due to rainfall (Kimaro et al, 2002). For estimating this process, the different land covers which cause spatial variability in the recharge rate needed to be taken into consideration.

The most suitable equation in terms of simplicity and minimum dataset requirements, although still reliable and accurate was the one described by Campbell (1985). Actually Campbell describes this as an equation for hydraulic conductivity but I use it in my thesis as an equation for recharge:

$$Recharge_{(t)} = SatHK\,(Moist_{(t)} / SatMoist)^{((2b)+3)} \tag{6-12}$$

$Recharge_{(t)}$ = ground water recharge at timestep t (mm water/hour)
SatHK = saturated hydraulic conductivity (mm water/hour)
$Moist_{(t)}$ = soil-moisture content at timestep t (mm water/mm soil)
SatMoist = saturated volumetric water content (constant) (mm water/mm soil)
b = decay constant of soil-moisture characteristics curve (dimensionless)

Such an equation was found not to work well with a timestep as long as the one used here since when multiplied by time very large volumes of water can be removed from the soil in a single timestep without the negative feedback of soil moisture reduction which further reduces the rate of recharge that would occur with a shorter timestep. Thus the recharge was calculated as the integral of fluxes taking account of soil moisture reduction within the timestep used by the model. The process of integration consisted of several steps:

1. The value of soil-moisture content is classified from 0 to 100% in increasing intervals of 10, and therefore, in total 10 soil moisture fractions will be calculated. The sum of all calculated fractions must be the total soil-moisture content.

$$\text{Moist}\ (z) = \text{Moist}_{(t)}\ (z / 100) \qquad (6\text{-}13)$$

Moist (z) = (z) percentage of the total soil moisture content (mm water/mm soil)
$\text{Moist}_{(t)}$ = soil-moisture content at timestep t (mm water/mm soil)
(z) = percentage fraction which increases from 0 to 100 in intervals of 10

2. Secondly, the groundwater recharge is calculated per each of the tenth soil moisture fractions, and therefore, the recharge is also per fraction.

$$\text{GRECH}\ (z) = \text{SatHK}\ ((\text{Moist}\ (z) / \text{SatMoist})^{((2\ b) + 3)}) \qquad (6\text{-}14)$$

GRECH (z) = recharge water (z) percentage of groundwater recharge related to the (z) fraction of soil moisture content (mm water/hour)
SatHK = saturated hydraulic conductivity (mm water/hour)
Moist (z) = (z) percentage of the total soil moisture content (mm water/mm soil)
SatMoist = saturated volumetric water content (constant) (mm water/mm soil)
b = decay constant of soil moisture characteristics (dimensionless)
(z) = percentage fraction which increases from 0 to 100 in intervals of 10

3. Calculation of the positive difference between the values of a moisture fraction and its subsequent one, in order to determine the change in soil moisture between fractions.

$$\text{Moist(z)change} = \text{Moist}\ (z) - \text{Moist}\ (z\text{-}10) \qquad (6\text{-}15)$$

Moist(z)change = soil moisture value of the difference between two consecutive soil moisture fractions (mm water/ mm soil)

Moist (z) = (z) percentage of the total soil moisture content (mm water/ mm soil)
(z) = percentage fraction which increases from 0 to 100 in intervals of 10

4. Soil moisture unit conversion from (mm water/mm soil) to mm water through soil depth (mm soil).

$$Moist(z)mm = (Moist(z)change) * Depth_{(t)} \quad (6\text{-}16)$$

Moist(z)mm = soil moisture change between fractions (mm water)
Moist(z)change = soil moisture value of the difference between two consecutive soil moisture fractions (mm water/mm soil)
Moist (z) = (z) percentage of the total soil moisture content (mm water/mm soil)
Depth $_{(t)}$ = soil depth at timestep t (mm soil)
(z) = percentage fraction which increases from 0 to 100 in intervals of 10

5. Calculation of the time needed for the water to recharge in every fraction determined.

$$RechTime(z) = (Moist(z)mm / GRECH(z)); \quad GRECH(z) ? 0 \quad (6\text{-}17)$$
$$RechTime(z) = 0; \quad GRECH(z) = 0$$

RechTime(z) = time needed for the water to recharge the (z) fraction of the total groundwater recharge (hour)
GRECH (z) = recharge water (z) percentage of groundwater recharge related to the (z) fraction of soil moisture content (mm water/hour)
Moist(z)mm = soil moisture change between fractions (mm water)
(z) = percentage fraction which increases from 0 to 100 in intervals of 10

6. Calculation of the total time needed for recharging the total amount of water (from all tenth recharge fractions) into the soil.

$$\text{RechTimeTotal} = \overset{10}{\underset{i=0}{?}} \ \text{ReachTime}\ (z+10i) \qquad (6\text{-}18)$$

RechTimeTotal = time for recharging the total water associated to each of the tenth recharge fractions (hour)

RechTime(z) = time needed for the water to recharge the (z) fraction of the total groundwater recharge (hour)

(z) = percentage fraction which increases from 0 to 100 in intervals of 10

7. Computation of the percentage from the total time needed for the overall water recharge, related to every of the tenth fractions of recharge times.

$$\text{RechFrac}(z) = (\text{RechTime}(z) / \text{RechTimeTotal}), \quad \textit{RechTime(z) ? 0} \qquad (6\text{-}19)$$

$$\text{RechFrac}(z) = 0, \quad \textit{RechTime}(z) = 0$$

RechFrac(z) = (z) fraction of the total time needed for the total water recharge (hour)

RechTimeTotal = time for recharging the total water associated to each of the tenth recharge fractions (hour)

RechTime(z) = time needed for the water to recharge the (z) fraction of the total groundwater recharge (hour)

(z) = percentage fraction which increases from 0 to 100 in intervals of 10

8. Calculation of the mm water per hour of each recharge fraction.

$$\text{Recharge}(z) = \text{GRECH}(z) * \text{RechFrac}(z) \qquad (6\text{-}20)$$

Recharge(z) = (z) fraction of the total groundwater recharge (mm water/ hour)

GRECH (z) = recharge water (z) percentage of groundwater recharge related to the (z) fraction of soil moisture content (mm water/ hour)

RechFrac(z) = (z) fraction of the total time needed for the total water recharge (hour)

(z) = percentage fraction which increases from 0 to 100 in intervals of 10

9. The total groundwater recharge is the sum of all the ten fractions:

$$GRECH = \sum_{i=0}^{10} Recharge\ (z+10i) \qquad (6\text{-}21)$$

GRECH = total groundwater recharge (mm water/hour)

Recharge(z) = (z) fraction of the total groundwater recharge (mm water/hour)

(z) = percentage fraction which increases from 0 to 100 in intervals of 10

10. Finally, since the timestep of the model is monthly these hourly results had to be converted into monthly/equivalents.

$$Recharge_{(t)} = GRECH_{(t)} * (v) * (w) \qquad (6\text{-}22)$$

$Recharge_{(t)}$ = total groundwater recharge at timestep t (mm water/month)

$GRECH_{(t)}$ = total groundwater recharge at timestep t (mm water/hour)

(v) = 24, constant related to the hours of a day

(w) = 31, constant related to the days of a month

#### 6.3.6.9. Soil erosion sub-model

The selected soil erosion model described by Thornes (1990), was considered the best for the DIS template. This model was developed for the Mediterranean region through the use of empirical plot scale data, and although it has been demonstrated to be reliable for these areas, it is very simple to operate and the input data requirements may be considered low.

$$EROS_{(t)} = K\ [(OFLOW_{(t)})^{M}]\ (S^{N})\ (e^{(-0.07\ Veg)});\quad \textit{EROS} = 0 \qquad (6\text{-}23)$$

$EROS_{(t)} = 0; \quad \mathit{EROS} < 0$

$EROS_{(t)} = Depth_{(t-1)}; \quad \mathit{EROS} > \mathit{Depth}$

$EROS_{(t)}$ = soil erosion at timestep t (mm soil/month)

K = soil erodability factor (constant) (dimensionless)

$OFLOW_{(t)}$ = overland flow computed from the routing of the runoff water generation at timestep t (mm water/month)

M = discharge scaling factor (constant) (dimensionless)

S = tangent of the slope map (degrees)

N = Slope constant (dimensionless)

Veg = vegetation cover (percentage)

$Depth_{(t-1)}$ = soil depth calculated in the previous t timestep (mm soil)

The first restriction to be applied to this equation accounts for the fact that the value of erosion cannot be negative, and the second one is that the value of soil erosion cannot be greater than the depth of the soil.

Soil depth has an important role in the overall DIS model, since it is part of several sub-models. Therefore, soil depth is considered a dynamic parameter, and its value progressively reduces when the soil is eroded.

$$Depth_{(t)} = Depth_{(t-1)} - EROS_{(t)}; \quad \mathit{Depth}_{(t-1)} > \mathit{EROS}_{(t)} \qquad (6\text{-}24)$$

$$Depth_{(t)} = 0.0001; \quad \mathit{Depth}_{(t-1)} = \mathit{EROS}_{(t)}$$

$EROS_{(t)}$ = soil erosion at timestep t (mm soil/ month)

$Depth_{(t)}$ = soil depth at timestep t (mm soil)

$Depth_{(t-1)}$ = soil depth calculated in the previous t timestep (mm soil)

Finally, for mathematical reasons, since soil depth acts as denominator in some equations, its value cannot be zero and therefore it would be set as 0.0001 in the case it is negative or zero.

### 6.3.7. INITIAL SELECTION OF PARAMETER VALUES FOR THE DIS

#### 6.3.7.1. Soil-moisture content sub-model

The input parameters required in the soil moisture content sub-model are sourced in four different ways: (a) fieldwork parameters such as: soil moisture content, bulk density and soil depth; (b) parameters calculated on the basis of fieldwork determinations, such as soil porosity; (c) parameters related to three other sub-model outcomes, such as: soil actual infiltration capacity (AINF), actual evapotranspiration (ActET) and groundwater recharge (Recharge); and (d) the land-use map (LUM) of the study area.

Soil moisture content (Moist) and bulk density (BD) values are the average of bi-seasonal field determinations, throughout the one-year study period, whereas soil depth (Depth) values correspond to a single determination. All fieldwork parameters are expressed for land use (Table 6.2.). Soil porosity (poros) was calculated from BD values and a particle density (2.25 $g/cm^3$), and therefore it is also expressed per land use (Table 6.2) (see chapter four). On the basis of the LUM, the maps of Moist, Depth and poros, were created and directly applied in the soil moisture content sub-model. Bulk density values were only indirectly used for soil porosity calculation. The values of the AINF, ActET and Recharge parameters were set up as zero in the first timestep of the model run, since these sub-model outcomes had not been yet calculated by the model.

Moist, Depth, AINF, ActET, and Recharge are dynamic parameters in the DIS template. However, an initial value is needed in the first timestep as they also act as input variables in the first sub-model, the soil moisture content. These inputs values were set up in the *initial* section of the PCRaster model script. From the second timestep on, their value in this sub-model will be the one calculated in the corresponding sub-models in the previous timestep. The rest of parameters such as soil porosity and land-use maps, were considered constant and were defined and named in the *binding* section.

Table 6.2. Fieldwork parameters and those calculated from them

| **Land Use** | **Moist (mmwater/mmsoil)** | **BD ($g/cm^3$)** | **Poros (mmpores/mmsoil)** | **Depth (mmsoil)** |
|---|---|---|---|---|
| Dense Cork Trees (DCT) | 0.152 | 1.029 | 0.543 | 290 |
| Sparse Cork Trees (SCT) | 0.111 | 1.250 | 0.444 | 150 |
| Dense Shrubs (DS) | 0.148 | 1.241 | 0.448 | 390 |
| Sparse Shrubs (SS) | 0.112 | 1.326 | 0.411 | 200 |
| Cultivated Olive Trees (COT) | 0.116 | 1.319 | 0.414 | 400 |
| Abandoned Olive Trees (AOT) | 0.114 | 1.178 | 0.476 | 250 |
| Cultivated Vineyards (CV) | 0.115 | 1.562 | 0.306 | 150 |
| Pine Trees Afforestation (PTA) | 0.084 | 1.033 | 0.541 | 100 |
| Roads (ROAD) | 0.001 | 0.001 | 0.001 | 0.001 |
| Bare Soil (BS) | 0.123 | 1.305 | 0.421 | 300 |

Moist: soil moisture content; BD: bulk density; Poros: soil porosity; Depth: soil depth.

For mathematical operability reasons, the land use related to the roads (ROAD), was set up with a Moist, Depth, BD, and poros value of 0.001, for all timesteps in the model.

#### 6.3.7.2. Potential infiltration capacity sub-model

Four input parameters were required in the potential infiltration capacity sub-model: the LUM, the saturated hydraulic conductivity (SatHK), the field potential infiltration capacity (Pot) and the Hortonian empirical constant (h) (Table 6.3).

The SatHK was calculated individually for each of the ten selected land uses on the basis of the HYDROPAR spreadsheet model (Mulligan, 1996). On the other hand, Pot values were obtained from fieldwork infiltration determinations and also per land use. Both, SatHK and Pot data were converted into maps through the LUM. These two parameters considered constants in the DIS, and were described and named in the *binding* section of the script. With reference to h, this constant was empirically determined on the basis of the Hortonian equation, which states an exponential relationship between soil moisture content and soil potential infiltration capacity.

Here, the initial infiltration rate corresponds to the maximum infiltration rate (Pot), when soil moisture content (Moist) is assumed to be negligible (zero). On the contrary, when the soil is saturated, soil moisture is assumed to equal the soil porosity, and then the potential infiltration capacity is assumed to equal the value of the saturated hydraulic conductivity (SatHK). Therefore, on the basis of the previous explanation, h was calculated for each land use and was found to be approximately 25 for all of them. The potential infiltration value related to the roads land use, was set up as zero per all timesteps, since this environment does not infiltrate and is important in the runoff and overland-flow sub-models.

Table 6.3. Potential infiltration sub-model input parameters

| **Land Use** | **SatHK (mmwater/hour)** | **Pot (mmwater/hour)** | **h (dimensionless)** |
|---|---|---|---|
| Dense Cork Trees (DCT) | 9.93 | 1132.524 | 25 |
| Sparse Cork Trees (SCT) | 7.21 | 2900.649 | 25 |
| Dense Shrubs (DS) | 6.228 | 2571.08 | 25 |
| Sparse Shrubs (SS) | 4.568 | 1776.197 | 25 |
| Cultivated Olive Trees (COT) | 3.83 | 401.217 | 25 |
| Abandoned Olive Trees (AOT) | 5.598 | 773.592 | 25 |
| Cultivated Vineyards (CV) | 3.432 | 588.128 | 25 |
| Pine Trees Afforestation (PTA) | 10.479 | 3447.103 | 25 |
| Roads (ROAD) | 0 | 0 | - |
| Bare Soil (BS) | 4.009 | 431.969 | 25 |

SatHK: saturated hydraulic conductivity; Pot: experimental potential infiltration capacity; h: empirical hortonian constant;

### 6.3.7.3. Runoff and Actual infiltration capacity sub-model

The input parameters required in the runoff sub-model were rainfall amount, rainfall intensity, rainfall frequency distribution, and potential infiltration capacity, which values were obtained from the previous sub-model calculations in the same timestep.

Table 6.4. Monthly rainfall amount related to three different yearly rainfall scenarios

| | **Rainfall (mm water)** | | |
|---|---|---|---|
| **Months** | **Dry (1980)** | **Normal (1997)** | **Wet (1993)** |
| January | 28.5 | 70.5 | 0.4 |
| February | 27.2 | 1 | 121 |
| March | 51.8 | 0 | 69 |
| April | 43.8 | 18.5 | 122 |
| May | 81.6 | 38 | 82 |
| June | 46 | 88.6 | 44 |
| July | 27 | 32.6 | 61 |
| August | 13.4 | 73.7 | 56.7 |
| September | 18.4 | 46 | 94.5 |
| October | 28 | 73.4 | 177.1 |
| November | 50.4 | 79.5 | 162.2 |
| December | 14.8 | 63.6 | 6 |

Dry: dry scenario (430.90 mm rain/year); Normal: normal scenario (585.40 mm rain/year); Wet: wet scenario (995.90 mm rain/year).

Since one of the aims of the model was to determine the role of rainfall on soil erosion and runoff generation, different scenarios of rainfall were considered for testing. From a rainfall record from 1945 to 2001 year, provided by the Figueres Automatic Weather Station, three years were selected on the basis of their total annual rainfall amount: (a) a wet year (1993), where the annual rainfall amount was higher than the annual average of these 50 years, (b) a normal or average rainfall year (1997), and (c) a dry year (1980), where the annual rainfall amount was lower than the average (Table 6.4).

Rainfall intensity and rainfall frequency distribution data were also available from the Figueres Automatic Weather Station and are listed in Table 6.5.

The input parameters required in the actual infiltration capacity sub-model are mainly the same as the ones needed in the runoff sub-model, since both sub-models are based on the same equation and therefore they are tightly interrelated.

Table 6.5. Rainfall intensity and rainfall frequency distribution data

| Rainfall Intensity (mmwater/hour) | Rainfall frequency distribution (%) |
|---|---|
| 1 | 14.5043 |
| 2 | 8.1171 |
| 3 | 9.7804 |
| 4 | 4.1251 |
| 5 | 2.1291 |
| 6 | 12.1756 |
| 8 | 2.0625 |
| 12 | 19.6939 |
| 18 | 0.3992 |
| 24 | 14.6374 |
| 36 | 6.4538 |
| 48 | 1.1976 |
| 60 | 1.3972 |
| 72 | 1.3972 |
| 84 | 0.2661 |
| 96 | 0.3992 |
| 108 | 0.3327 |
| 132 | 0.0665 |
| 144 | 0.2661 |
| 156 | 0.1331 |
| 180 | 0.0665 |
| 192 | 0.1996 |
| 204 | 0.0665 |
| 264 | 0.0665 |
| 276 | 0.0665 |

#### 6.3.7.4. Overland flow sub-model

The direct input parameters in the overland flow sub-model were the local drainage directions map (LDD), and the outcomes from the actual infiltration capacity (AINF) and runoff (Runoff) sub-models previously calculated in the same timestep. Indirectly the digital elevation model of the study area (DEM), was also required as input in this sub-model, since it was the basis for creating the LDD.

### 6.3.7.5. Potential evapotranspiration and Actual evapotranspiration sub-models

The input parameters in the potential evapotranspiration sub-model are: digital elevation model (DEM), land use map (LUM), monthly temperature (Temp), solar radiation at the land surface (SRLS) and also at the top of the atmosphere (SRTA), albedo, and some constants such as: latent heat of vaporization (Lambda), Priestley and Taylor constant (Ptcoeff), and the psychrometric constant (Psychro).

Table 6.6. Monthly temperature and solar radiation data

| | Temperature (ºC) | | | Solar Radiation (MJ/m$^2$) | |
|---|---|---|---|---|---|
| **Months** | **Dry (1980)** | **Normal (1997)** | **Wet (1993)** | **SRLS (AWS)** | **SRTA (Solargen)** |
| January | 6.8 | 8.75 | 8.4 | 6.6 | 15.16 |
| February | 10.3 | 10.8 | 8.55 | 10.8 | 20.39 |
| March | 10.45 | 12.7 | 10.7 | 14.8 | 27.61 |
| April | 13.1 | 15.1 | 13.3 | 17.3 | 34.60 |
| May | 15.1 | 18 | 17.5 | 18.2 | 39.62 |
| June | 18.7 | 21.7 | 22.2 | 20.8 | 41.64 |
| July | 22.43 | 23 | 23 | 22.2 | 40.70 |
| August | 22.35 | 23.8 | 24.35 | 19.2 | 36.73 |
| September | 21.15 | 21.05 | 19.85 | 15.6 | 30.48 |
| October | 15.35 | 18.2 | 15.5 | 10.3 | 23.17 |
| November | 9.3 | 11.85 | 10.8 | 7.4 | 16.78 |
| December | 6.9 | 9.05 | 9.1 | 5.9 | 13.68 |

Dry: dry scenario (430.90 mm rain/year); Normal: normal scenario (585.40 mm rain/year); Wet: wet scenario (995.90 mm rain/year); SRLS: monthly daily average solar radiation at the land surface; AWS: Automatic Weather Station; SRTA: monthly daily average solar radiation at the top of the atmosphere.

Temperature data were selected according to the three rainfall scenarios (Dry, Normal and Wet), and therefore three monthly dataset were used (Table 6.6). Regarding the solar radiation data, two different kinds were required, the SRLS, available from the AWS and the SRTA, calculated on the basis of the Solargen model (Mulligan, 1996) (Table 6.6). In addition, the DEM was also needed for creating the SRTA maps. Albedo values referred to each of land use were also required (Table 6.7). These data were determined on the basis of literature review (Allen et al., 1998), and although they can vary widely with the time of the day, season, latitude and cloud cover, they were considered constant. The albedo map was created on the basis of the LUM. The solar radiation data together with the determined albedo map were used for calculating the monthly net solar radiation. Lambda was set up as 2.45 MJ/kg, which corresponds to a temperature of 20ºC. The values of P-T and Psy constants are listed in Table 6.7.

Table 6.7. Input constants in the potential evapotranspiration sub-model

| Land Use | Albedo | Constants | Values |
|---|---|---|---|
| Dense Cork Trees (DCT) | 0.16 | | |
| Sparse Cork Trees (SCT) | 0.17 | Latent heat vaporization (Lambda)(MJ/kg) | 2.45 |
| Dense Shrubs (DS) | 0.24 | | |
| Sparse Shrubs (SS) | 0.23 | | |
| Cultivated Olive Trees (COT) | 0.17 | Priestley-Taylor (P-T) (dimensionless) | 1.28 |
| Abandoned Olive Trees (AOT) | 0.18 | | |
| Cultivated Vineyards (CV) | 0.15 | | |
| Pine Trees Afforestation (PTA) | 0.16 | Psychrometric (Psychro) (KPa/ºC) | 1.26 |
| Roads (ROAD) | 1 | | |
| Bare Soil (BS) | 0.15 | | |

With reference to the actual evapotranspiration sub-model, it only required data from the soil moisture content and the potential evapotranspiration sub-models.

#### 6.3.7.6. Groundwater-recharge sub-model

The groundwater-recharge sub-model required the calculation of the saturated hydraulic conductivity (SatHK) and the b, a constant calculated as a function of soil texture. Both parameters were estimated per land use by the HYDROPAR model (Mulligan, 1996) (Table 6.8), and the correspondent map was created on the basis of the land use map (LUM) of the study area. In addition, the saturated volumetric water parameter (SatMoist) was also needed in the equation and was set as constant with a value of 1 mm water/mm soil. The soil moisture content (Moist), outcome of the first sub-model (same timestep), was also needed for the overall process calculation.

Table 6.8. Saturated hydraulic conductivity and b parameter

| Land Use | b (dimensionless) | SatHK (mm water/hour) |
|---|---|---|
| Dense Cork Trees (DCT) | 2.126 | 9.93 |
| Sparse Cork Trees (SCT) | 1.844 | 7.21 |
| Dense Shrubs (DS) | 1.886 | 6.228 |
| Sparse Shrubs (SS) | 1.692 | 4.568 |
| Cultivated Olive Trees (COT) | 2.088 | 3.83 |
| Abandoned Olive Trees (AOT) | 1.896 | 5.598 |
| Cultivated Vineyards (CV) | 1.791 | 3.432 |
| Pine Trees Afforestation (PTA) | 2.231 | 10.479 |
| Roads (ROAD) | 0 | 0 |
| Bare Soil (BS) | 1.836 | 4.009 |

SatHK: saturated hydraulic conductivity.

Once again, for conceptual reasons as well as mathematical, the value of the SatHK and b, were set to zero for the "Roads" land use.

#### 6.3.7.7. Soil erosion sub-model

According to Zhang (1999), the coefficients of the soil erosion model are commonly identified by field measurements based on rainfall events and they are considered to be very changeable from one environment to another. However, it has been noticed by Thornes (1985, 1990) that all these coefficients have their thresholds. Here, for the maximum sediment transport capacity and vegetation protection, the values for N, M and b, might be 2, 1.66 and –0.07 respectively, although further research is still needed.

The soil erosion equation accounts for several constants such as: the erodability factor (K = 2), the discharge scaling factor (M = 1.66) and the slope constant (N = 2) (Thornes, 1990). Additionally, the vegetation cover (Veg), soil depth (Depth) slope map (Slopedeg) and overland flow (OFLOW) values were also needed. Veg and Depth data (Table 6.9) were obtained per land use from fieldwork determinations and the related maps were created through the land use map (LUM). On the other hand, the slope map was directly derived from the digital elevation map (DEM) and the OFLOW values were those previously calculated in the overland flow sub-model (same timesptep).

For mathematical reasons, in order to achieve a good performance of the model, the “roads” land use was set up with a vegetation cover and soil depth value of “0.001”.

Table 6.9. Vegetation cover and soil depth data per land use

| Land Use | Vegetation cover (%) | Soil depth (mm soil) |
|---|---|---|
| Dense Cork Trees (DCT) | 80 | 290 |
| Sparse Cork Trees (SCT) | 50 | 150 |
| Dense Shrubs (DS) | 70 | 390 |
| Sparse Shrubs (SS) | 55 | 200 |
| Cultivated Olive Trees (COT) | 25 | 400 |
| Abandoned Olive Trees (AOT) | 75 | 250 |
| Cultivated Vineyards (CV) | 20 | 150 |
| Pine Trees Afforestation (PTA) | 79 | 100 |
| Roads (ROAD) | 0.001 | 0.001 |
| Bare Soil (BS) | 15 | 300 |

### 6.3.8. VERIFICATION PROCESS OF THE DIS MODEL

The run of the DIS model was carried out once the model script was finished. However, before the model run completely for the 12 set up timesteps, the programme produced several errors messages. This first set of errors was identified as “writing mistakes”, and therefore, they were easily and directly corrected in the script.

Afterwards, the verification process aimed at ascertaining possible conceptual errors in the model by looking at the outcomes of the model. Hence, the first step was to convert the outputs of the model, from spatial to text format for their easier analysis. This was achieved through the use of several PCRaster operators such as: *maptotal*, *areamap*, timeoutput. In addition, the temporal scale of the model set up as 12 timesteps (1 year), was extended to 120 timesteps (30 years), in order to ascertain the good performance of the model, through testing its regularity and performance over time. Then, data in text format was converted into Excel and the appropriate analysis was conducted assessing whether the results were logically correct or not. Throughout this process a few errors of conceptual nature were found in the DIS model, and were mostly corrected by finding abetter fit or adjusting some of the sub-models equations (see section 5.3.6).

The complete script of the DIS model including the verification process operations is presented in the appendix 4. Chapter seven is completely dedicated to the sensitivity analysis of the DIS and the validation process will be widely detailed in chapter eight, together with the generation of the scenarios outcomes.

# 7. SENSITIVITY ANALYSIS OF THE DIS MODEL



## 7.1. SENSITIVITY ANALYSIS: A GENERAL APPROACH

Sensitivity analysis is a crucial part of the development of any kind of model. Essentially it is the process of varying model input parameters over a reasonable range and observing the relative change in model response. Sensitivity analysis can be used for several purposes:

1. To demonstrate the sensitivity of the model simulations to uncertainty in values of model input data. As a result the sensitivity of one model parameter relative to other parameters can be demonstrated.

2. To determine the direction of future data-collection activities. Data for which the model is relatively sensitive would require future characterization, as opposed to data for which the model is relatively insensitive (Cullen and Frey, 1999).

3. In the case of risk models, it will be useful to identify the most significant exposure or risk factors and aid in developing priorities for risk mitigation (e.g., Baker et al., 1999).

4. It can play an important role in model verification and validation throughout the course of model development and refinement (e.g., Kleijnen, 1995; Kleijnen and Sargent, 2000; and Fraedrich and Goldberg, 2000). In more detail, it can be pointed out that:

    4.1. It can be helpful in verification in terms that if a model responds in an unacceptable way to changes in one or more inputs, then troubleshooting efforts can be focused on identifying the source of the problem (Frey and Patil, 2001).

    4.2. In reference to model validation it can be useful in terms of helping to develop a “comfort level” with a particular model. If the model response is reasonable from an intuitive or theoretical perspective, then the model users may have some comfort with the qualitative behaviour of the model even if the quantitative precision or accuracy is unknown (Frey and Patil, 2001).

5. It can be used in model extrapolation in order to reveal how the model performs when the model is extrapolated (Frey and Patil, 2001).

6. To provide insight into the robustness of model results when making decisions (e.g., Phillips et al., 2000; Ward and Carpenter, 1996; Limat et al., 2000; Manheim, 1998; and Saltelli et al., 2000).

7. To apply in various fields including complex engineering systems, economics, physics, social sciences, environmental issues, medical decision making, and others (e.g., Oh and Yang, 2000; Baniotopoulos, 1991; Helton and Breeding, 1993; Cheng, 1991; Beck et al., 1997; Agro et al., 1997; Kewley et al., 2000; Merz et al., 1992).

Hence, these aforementioned targets make obviously clear and totally justified the performance of the sensitivity analysis to our particular model.

## 7.2. MATHEMATICAL METHODS OF SENSITIVITY ANALYSIS

There exist different types of sensitivity analysis techniques, from mathematical, and statistical, to geographical, and so on. Here, the focus is on the sensitivity analysis methods applied in addition to the fundamental modelling technique, and in particular on the mathematical methods for sensitivity analysis. These methods assess sensitivity of a model output to the range of variation of an input, and they typically involve calculating the output for a few values of an input that represent the possible range of the input (e.g., Salehi et al., 2000). They can evaluate the impact of range of variation in the input values on the output. Therefore, not only can they be used for identifying inputs that require further data acquisition or research (e.g., Ariens et al., 2001), but they can also be helpful in screening the most important inputs (e.g., Brun et al., 2001) as well as useful on the model verification and validation (e.g., Wotawa et al., 1997).

### 7.2.1. NOMINAL RANGE SENSITIVITY

The analysis used for testing the sensitivity of the Desertification Indicator System (DIS) is based on the mathematical method called *Nominal Range Sensitivity.* This method assesses the effect on model outputs exerted by individually varying only one of the model inputs across its entire range of reasonable values, while holding all other inputs at their nominal (initial) values (Cullen and Frey, 1999). The difference in the model output due to the change in the input variable is referred to as the *sensitivity* or *swing weight* of the model to that particular input variable (Morgan and Henrion, 1990). The sensitivity is represented as a positive or negative percentage change compared to the nominal solution. The sensitivity analysis can be repeated for any number of individual model inputs. This method allows the identification of the most relevant inputs to propagate through a model in a probabilistic framework (Cullen and Frey, 1999) as well as be used to prioritise data collection needs (Salehi el al., 2000).

This particular method is most reliable and suitable when applied to a linear model. In this case, it would be possible to rank the order of relative importance of each input based upon the magnitude of the calculated sensitivity measure as long as the ranges assigned to each sensitive input are accurate. However, for a non-linear model, the sensitivity of the output to a given input may depend on interactions with other inputs, which are not considered.

The main advantage of the Nominal Range Sensitivity method is its relative simplicity and easy application. If there are no significant interactions among the inputs and if ranges are properly specified for each input, its results are used to rank order key inputs. On the other hand, the main disadvantage is that for non-linear models, such as the present model of interest, it is more difficult to provide a reliable rank ordering of key inputs.

## 7.3. OBJECTIVES OF THE SENSITIVITY ANALYSIS OF THE DIS

A complete sensitivity analysis of the DIS model has been conducted in order to achieve the following aims:

(a) Taking into account the several sub-models composing the DIS template, the most remarkable ones with regard to their role as indicators of desertification in the target Mediterranean area, will be chosen as relevant outputs to be analysed for sensitivity.

(b) To select the most important inputs of the chosen outputs factors (sub-models) on the basis of their conceptual relevance in the model run. The aim will be to assess the sensitivity of the outputs to changes in these inputs factors. Three rainfall scenarios with different mean annual rainfall amount will be considered for the analysis.

(c) To evaluate the effects of the Dry, Normal and Wet rainfall scenarios on the overall model sensitivity, performing the same input changes and testing them to the same selected outputs of the model. This will allow us to study the significance of the rainfall data on the model run.

(d) To ascertain the key input factors in terms of their sensitivity to the selected output factors, as well as the ones presenting substantial unresponsiveness to the model.

(e) To rank the overall tested parameters according to their sensitivity to the model. The most sensitive ones will be regarded as the most important ones in terms of their contribution to the outcomes of the model, and for instance more accurate and precise data will be required if available and necessary. On the contrary, the most insensitive will be considered of minimum relevance to the model results and they could be omitted. The ranking of the most sensitive factors of the DIS model, should allow the identification of key parameters with regard to the main indicators of desertification of this particular template.

(f) To establish the sensitivity and stability of the whole model and its entire set of components.

## 7.4. COMPUTATION OF THE DIS MODEL SENSITIVITY

### 7.4.1. SELECTING THE INPUTS AND OUTPUTS OF THE DIS MODEL FOR THE SENSITIVITY ANALYSIS

As described in chapter six, the DIS is a spatio-numerical model composed by several interlinked main sub-models: soil moisture content (SM), potential infiltration capacity (PI), runoff generation (R), actual infiltration capacity (AI), overland flow (cell based runoff) (OF), potential evapotranspiration (PET), actual evapotranspiration (AET), groundwater recharge (GR), and soil erosion (SE).

From the aforementioned nine sub-models and taking into account their relevance to land degradation and desertification, three of them have been selected as outputs on the sensitivity testing: the M, OF, and SE sub-model.

In reference to the input variables to be tested, the most remarkable ones in terms of meaning and conception to the model have been chosen from all the sub-models. Eleven input variables have been selected and then classified into two main groups in accordance to their role on the overall model. The first group of parameters concerns *Soil Erosion parameters* (SEP), and is composed by: soil depth (SD), soil erodibility (K), discharge scaling factor (M), slope constant (N), and vegetation cover (Veg). The second group addresses *Hydrological parameters* (HYP) such as: soil porosity (poros), saturated hydraulic conductivity (SatHK), Priestley-Taylor constant (PT), psychrometric constant (Psy), Hortonian constant (h) and b values (bvalues).

### 7.4.2. PROCEDURE OF SENSITIVITY ANALYSIS OF THE DIS MODEL

The sensitivity analysis of the DIS model has been conducted three separate times with respect to the different rainfall scenarios of interest. However, the same computational procedure has been equally performed per each scenario, being the only difference, the monthly precipitation data used in the sensitivity runs of the DIS model in every rainfall scenario. Total annual rainfall amount increases from the Dry (430.90 mm) to the Normal (585.40 mm) and Wet years (995.90 mm), respectively.

The initial value of each selected input factor will be increased and decreased a 10% in an accumulative way, covering a rank ranging from –100% to +100% of the unperturbed value. Therefore, the sensitivity analysis concerning a particular input factor was performed twenty different times, taking into account the full range of change of the input. A specific batch file was created for the sensitivity testing of each of the eleven input factors in order to reduce the time consuming performance needed for the 20 runs of each input. The overall set of input factors required 220 runs of the DIS model. Furthermore, taking into consideration the sensitivity of the model under three different rainfall scenarios, a total of 660 runs were necessary.

Once all the runs of the model were conducted, the whole sensitivity to the input factors in the model was computed from the model output using a spreadsheet model, which calculated the change in the output factors per unit change in the input factors. The equation used for this purpose is:

$$S = dO\% / dI\% \tag{7-1}$$

I% = factor value expressed as percentage change from the unperturbed value.
O% = output value expressed as percentage change from the unperturbed value.

Every single sensitivity value is computed for all the selected outputs, addressing the entire range of factor changes. Sensitivity results are represented in a series of 3-D bar charts. Every plot is expressed by the same scale ranging from –2 to +2, where –2 is two units of decrease in the response factor (output) per unit of change in the input factor, and +2 is two units of increase in the response factor per unit of change in the input factor. For every illustration the *X* axis describes the input factors, which are presented in four classes addressing four different range groups: from –100% to –50%, from –50% to 0% (0%: unperturbed value or control point) (negative ranges), from 0% to +50% and from +50% to +100% (positive ranges). The *Z* axis presents model sensitivity as height of the bars of the chart, where the positive or negative sign of sensitivity shows the direction of the response. The different model output variables are represented in the *Y* axis of the diagrams, making the relative sensitivity of different system components clear and easy to understand.

Concisely, the sensitivity testing will mainly focus on: (a) the sensitivity to soil erosion parameters; (b) the sensitivity to hydrological parameters; and to some extent (c) the sensitivity to precipitation.

## 7.5. RESULTS OF THE SENTISIVITY ANALYSIS OF THE DIS

### 7.5.1. GENERAL OBSERVATIONS

Some general observations regarding the sensitivity of the DIS model may be drawn, although, the complexity of interactions within the processes composing the model is difficult to tackle. From illustrations 7.1 to 7.6, some important remarks can be highlighted:

1. Different output factors (soil moisture content, overland flow and soil erosion) have a dissimilar sensitive response to the exact change in percentage in a single input factor (e.g. soil depth or saturated hydraulic conductivity).

2. In general, each specific output factor is characterized by a different sensitivity level to changes in distinct input factors. For example, soil erosion, overland flow and soil moisture content are remarkably different in terms of their sensitivity to changes in saturated hydraulic conductivity.

3. The spatio-numerical model clearly shows a different sensitivity across all range of changes of a particular input factor, accounting for the non-linear characteristics of the processes describing every sub-model of the DIS. It is very likely that the model presents a relatively insensitivity with regards to a precise range of change of a individual input factor as opposed to a relatively high sensitivity of the same input but concerning another range of change.

4. In general, figures illustrate a similar pattern with regard to the sensitivity of the output factors to the change of every input factor for the three rainfall scenarios. This similar trend for the Dry, Normal and Wet rainfall, is stronger when concerning the hydrological parameters than the ones related to soil erosion. However, the percentage of change of the inputs addressing the relative sensitivity or insensitivity of the outputs is by no means the exact absolute value in any case or scenario.

5. It is clear from the analysed graphs that sometimes, the overall set of outputs is unresponsive to a particular input accounting for its irrelevance to the entire DIS model. For example, regarding the analysis of the hydrological variables it is apparent that the three outputs are totally insensitive to changes in the b input.

### 7.5.2. SENSITIVITY RELATED TO SOIL EROSION PARAMETERS (SEP)

Figures 7.1, 7.2, and 7.3 illustrate the sensitivity of soil moisture content, overland flow and soil erosion to changes in parameters related to soil erosion (SEP) such as: soil depth, erodibility constant, discharge scaling factor, slope constant and vegetation cover, for the Dry, Normal and Wet rainfall scenarios respectively. These parameters of the DIS model have been grouped into the SEP category since they are either components of the soil erosion sub-model or they are directly related to it.

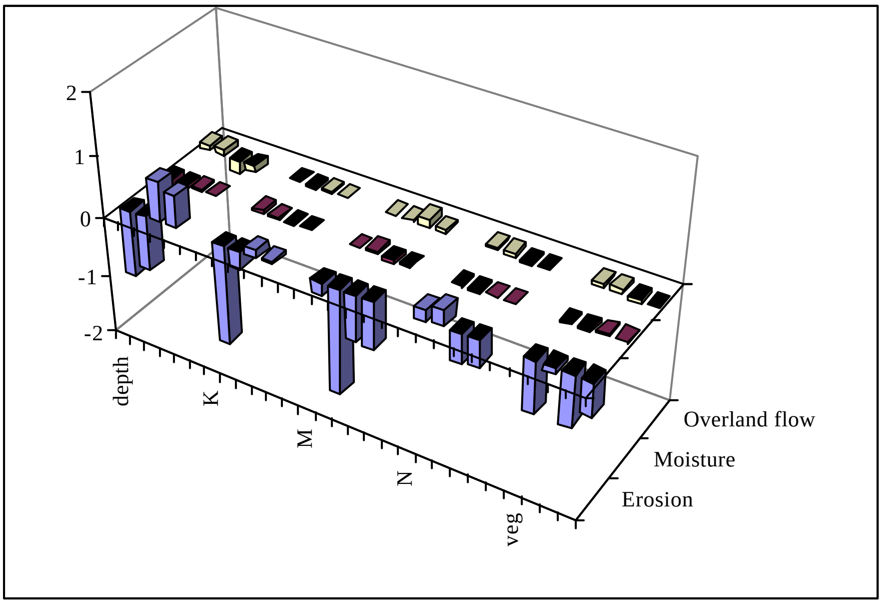


Figure. 7.1. Dry rainfall scenario. Sensitivity of soil erosion, soil moisture content, and overland flow to changes in soil erosion related parameters (SEP)

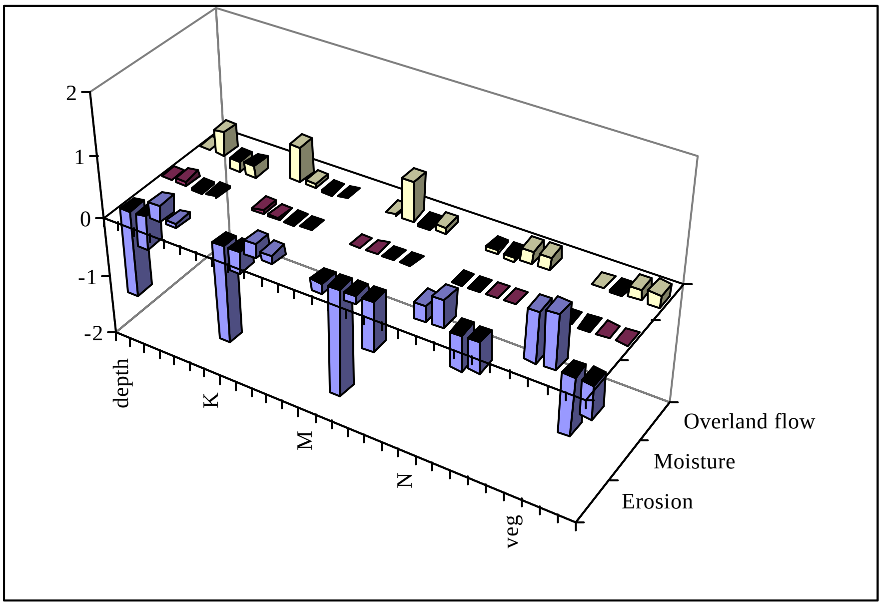


Figure 7.2. Normal rainfall scenario. Sensitivity of soil erosion, soil moisture content, and overland flow to changes in soil erosion related parameters (SEP)

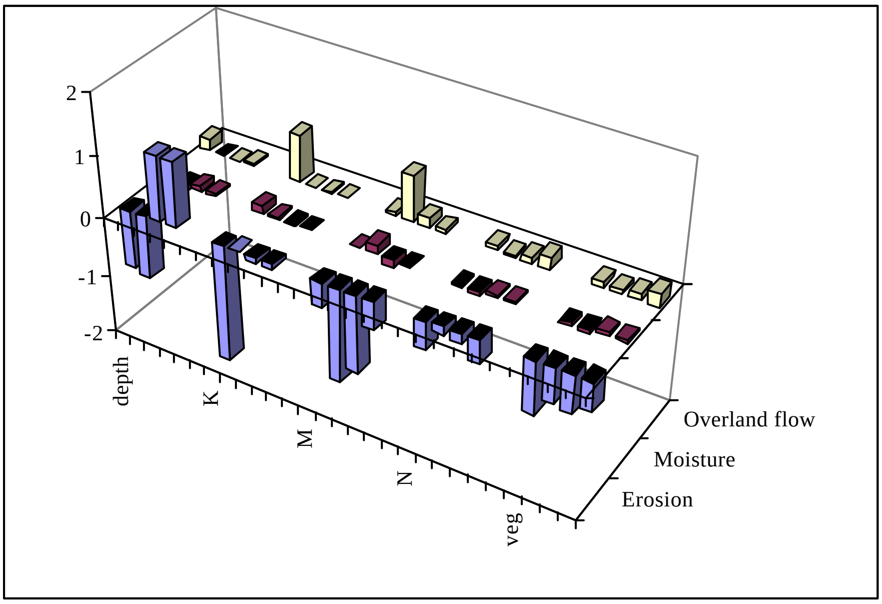


Figure 7.3. Wet rainfall scenario. Sensitivity of soil erosion, soil moisture content, and overland flow to changes in soil erosion related parameters (SEP)

### 7.5.2.1. Overall patterns

#### 7.5.2.1.1. Soil moisture content sensitivity to changes in SEP

Soil moisture content factor (SM), shows either total or very high unresponsiveness to changes in the entire set of soil-erosion parameters (SEP). Hence, in general all SEP suggest negligible sensitivity of this output. This group of selected parameters form part of the soil erosion sub-model in the DIS template. Consequently, it may be suggested that soil-erosion parameters are neither directly or indirectly related to the soil moisture content sub-model. In this context, the processes controlling soil moisture content as described in the model are: actual infiltration capacity (AI), actual evapotranspiration (AE), groundwater recharge (GR) as well as soil moisture content of the previous timestep (M). Therefore, it is reasonable to assume that soil moisture content is by no means influenced by soil erosion and for instance it does not depend on its fluctuation, explaining why it is not sensitive to SEP changes. This specific pattern can be verified for the three rainfall scenarios as showed in Figures 7.1 (Dry), 7.2 (Normal) and 7.3 (Wet), suggesting that the rainfall amount is not significant when testing soil moisture content sensitivity to changes on SEP.

#### 7.5.2.1.2. Soil-erosion sensitivity to changes in SEP

As was expected, soil-erosion output factor (SE) is suggested to be considerable sensitive to changes in the overall group of SEP. As mentioned above, SEP are directly related to soil erosion, since they are forming part of the mathematical equation, which expresses this process, in the DIS model. The substantial positive or negative sensitivity in terms of magnitude is clear per all SEP as well as per all rainfall scenarios as presented in Figures 7.1 to 7.3. However, comparing the three precipitation scenarios with regard to SE sensitivity to changes in a particular SEP, it is shown that in some cases the pattern of sensitivity varies greatly across the whole range of changes of a particular input factor. Conversely, some specific input factors present similar behaviour in every scenario and range of change. This will be explained in more detail later in this section.

#### 7.5.2.1.3. Overland-flow sensitivity to changes in SEP

The sensitivity of overland flow (OF) to changes in SEP, as presented in illustrations 7.1 to 7.3 for the different rainfall scenarios, is the most irregular one, in comparison to the soil-moisture content and soil erosion response factors. There exist notable differences on the OF sensitivity or insensitivity when analysing the several SEP individually. Thus, the input parameters showing significant OF sensitivity, at least in one of their four range of change are: the soil depth (depth), soil erodibility factor (K) and discharge scaling factor (M). On the other hand, the most insensitive input factors seem to be the slope constant (N) and the vegetation cover (veg). The contribution of every input factor to the sensitivity of the OF will be explained in more detail later in this chapter. Comparing the three charts, it is clear that the one related to the Normal rainfall scenario shows the highest insensitivity of the OF concerning the entire set of SEP. Particularly, the OF sensitivity in this scenario could be considered negligible. The suggested next less sensitivity scenario is the Wet and then the Dry. Despite the fact that conceptually the overland flow and soil erosion are two processes tightly related, the overland flow sub-model is totally independent of the soil erosion one, in the DIS model. The overland flow sub-model is directly affected by the actual infiltration capacity and the local drainage of the catchment and indirectly by the rainfall amount. Therefore, it was expected a contribution not particularly important of the SEP to the sensitivity of the overland flow, even though that with regard to the opposite situation, the soil erosion sub-model is directly dependent on the overland flow.

#### 7.5.2.2. Soil-erosion parameters

##### 7.5.2.2.1. Soil depth (depth)

Soil depth is considered an essential factor in the DIS model framework. Soil depth most important role is in the soil erosion sub-model, where it acts as a restriction, limiting the maximum potential amount of soil to be eroded. It is obvious that it is materially impossible that the amount of soil potentially erodible exceeds the one that it is actually composing the soil profile. In this same context, it is perceptible from the charts (Figures 7.1-7.3), that the decrease in soil depth values is directly proportional to the decrease in soil erosion and vice versa, as stated in the DIS model. So, it is comprehensible that soil-erosion sensitivity to changes in soil depth values is very high. This pattern is shown in all rainfall scenarios, although it is more apparent when the rainfall amount is higher. Therefore, the sensitivity of soil erosion to soil depth slightly increases from the Dry, Normal to the Wet scenario. To some extent, this could be explained on the basis that the more precipitation the wetter the soil, and for instance occurrence of overland flow is more likely, with the consequent detachment of soil particles resulting in soil erosion.

Despite the fact that, soil depth in the DIS model also influences the soil moisture sub-model in an indirect way, through converting the units of its components (AI, AET, and GR) from mmwater to $m^3$water/$m^3$soil, it is clear from the three figures that its contribution to the sensitivity of this output is completely negligible for all the three rainfall scenarios of study. In the overall model development, soil moisture calculation is not based on any expression describing this particular process, which would probably involve data from soil depth, bulk density and soil porosity. In this specific case, soil moisture sensitivity to soil depth would be expected to be much more significant. On the contrary, soil moisture in the DIS model is computed as the combination of several sub-models through their summation or/and subtraction closing the soil-water balance of which the spatio-numerical model was constructed. Therefore, from a mathematical point of view, it is not surprising that with regards to the design and operation of the model, soil-moisture output is not sensitive at all to changes in soil depth. This is verified in all rainfall scenarios.

In relation to the overland flow output factor it is observed that in terms of sensitivity to changes in soil depth it is slightly significant in comparison to soil moisture output. It is visible from the three diagrams that for some of the four ranges of change of the soil depth, the sensitivity of overland flow is perfectly negligible (e.g. the first range of change (-100% to –50%) in the Dry scenario). For instance, in the Wet rainfall scenario, overland flow only shows significant sensitivity to the first range of change but negligible to the remaining three. Irrespective of this, it seems that the sensitivity of overland flow follows slightly the opposite pattern than the sensitivity of soil erosion output. Moreover, in terms of absolute sensitivity when comparing both output factors, the one related to overland flows is presented as much lower per all rainfall scenarios. To some extent, the different sensitivity pattern showed in overland flow and soil erosion, and for instance the indirect relationship between soil depth and overland flow output could be explained through the assumption that when rainfall occurs, the shallower the soil, the less amount of water is likely to enter the soil (infiltration) and consequently an increase in the runoff generation is expected, and vice versa. It seems clear from the graphs that overland-flow sensitivity is slightly more important in terms of magnitude in the Dry rainfall scenario, followed by the Normal and then the Wet. Hence, the more rainfall amount the less sensitivity of overland flow to changes in soil depth input, accounting for the relevance of the climatic conditions addressing this specific analysis.

#### 7.5.2.2.2. Erodibility factor (K)

Soil erodibility factor (K) describes the inherent susceptibility of soil to be lost to erosion. In the DIS model, as explained in more detail in chapter six, the selected soil-erosion equation was the one described by Thornes (1990), which is specific for soils under Mediterranean conditions in terms of climatic, hydrologic and edaphic characteristics, as the ones related to the study area. K is set as a multiplying factor in the overall soil erosion mathematical expression. Normally, its calculation is based on some physico-chemical characteristics of the soil, such as texture, organic matter content, structure and permeability of the soil. However for simplicity reasons, in this selected equation, the K factor is assumed to have a constant value of 2.

In general, the sensitivity of soil erosion to changes in the K factor is relatively relevant. It seems from the graphs that soil erosion sensitivity tends to experiment a considerable decrease from the first negative range of change of the K factor (-100% to –50%) to the remaining three analysed ones: -50% to 0%; 0% to +50% and +50% to +100% respectively. Hence, soil erosion output shows its greatest sensitivity when assessing the maximum negative range (-100% to –50%) in the K factor value. This is confirmed for all rainfall scenarios. On the other hand, it seems that the amount of rainfall is also important in this testing, since the three diagrams suggest a noticeable decrease of the soil erosion sensitivity from the Dry to the Wet rainfall scenario. As a result, it can be concluded that the maximum sensitivity of the soil-erosion estimate to changes in K factor is mainly concentrated in the first range of change, followed by the second one, but only for the Dry and Normal scenario, whereas for the rest of ranges and all scenarios, soil erosion sensitivity could be considered negligible. Conceptually it would be logical to assume that the higher the K factor value, the more erodible the soil and consequently, the more intensive will be the associated soil erosion process. From the previous statement, it would be expected that the contribution of the K factor on the sensitivity analysis of the soil erosion output would be remarkably important. However, this analysis illustrates that the K factor is not as important in the soil erosion sensitivity analysis as it was expected.

In the DIS model, the K factor is neither mathematically nor conceptually associated to the soil-moisture content sub-model, so the sensitivity of this output factor with regards to changes in this input is believed insignificant. This is also true for all the rainfall scenarios as showed by the respective illustrations under evaluation.

With reference to the sensitivity of the overland flow output to changes in the K factor, very little significance is presented in all the three charts addressing the different rainfall scenarios. For instance, sensitive results are only shown for the first range of change (-100% to –50%) and only in the Dry and Wet rainfall scenarios. Because K factor is an intrinsic feature of the soil, it is quite understandable that this parameter affects particularly soil erosion above all the other outputs. Therefore, in this case, due to the indirect interrelationship between soil erodability and overland flow, the little sensitivity showed in this output, might be more associated to the variation on the monthly rainfall amount.

#### 7.5.2.2.3. Discharge scaling factor (M)

Roughness coefficients represent the resistance to flood flows in channel and flood plains. In particular, the discharge scaling factor (M) accounts for the resistance to flow presented by the channel. Appropriate values for M are typically estimated on the basis of tables developed through empirical study. Here, the value of M is defined by an empirically constant specially designed for the soil erosion model developed by Thornes (1990) and for instance adapted to Mediterranean conditions. Accordingly, the M coefficient is set as an exponential value of the overland flow component in the equation of soil erosion in the DIS framework. In theory, higher M values correspond to rougher channels (rocks, rigid vegetation, and woody debris) and lower values correspond to channels with smoother boundary materials and lower sinuosity. Conceptually it should be reasonable to say that the higher the M value, the slower the water will flow down (overland flow) which means that the soil particles will be less probable to be moved and for instance to be eroded. From these ideas, M value should be inversely proportional to soil erosion processes. Apparently, from the three graphs associated to the rainfall scenarios this is not completely veritable. On the contrary, although soil-erosion sensitivity to changes in M is presented as quite important, the pattern is always negative per all ranges of change of the input factor as well as per all the rainfall scenarios. Looking at the soil erosion equation in the DIS sub-model, from a mathematical point of view, the relationship between the M coefficient and soil erosion output should be totally opposed as the one reasoned above, since in the erosion sub-model, the M is the exponent factor of the overland flow which is a multiplying component of this expression. As a result, when the M value increases, so increases the overland flow and also the soil erosion outcome, and vice versa. Therefore, the pattern that follows the soil erosion sensitivity to changes in M coefficient is not what it should expected. What is quite clear is that per all the rainfall scenarios and ranges of change of the M values, soil erosion sensitivity always experiments a notable decrease.

Regarding the sensitivity of the soil moisture output factor to changes in M value, it can be contemplated as unresponsive for all the rainfall environments. On the one hand, this could be explained on the basis that the M is a specific and intrinsic parameter of the soil erosion sub-model, and therefore it does not have any direct or indirect influence to the soil moisture content sub-model in the DIS. On the other hand, from a conceptually point of view, there is no relationship between the roughness coefficient and the soil moisture content of the soil. So, the M input factor has a null contribution to the soil moisture content sensitivity.

The last output to be assessed in terms of sensitivity to changes in the M value is the overland flow. As previously mentioned, the M coefficient is only directly related to the soil erosion sub-model, but not to the overland flow. Accordingly, it should be logical not to expect an important sensitivity of the overland flow to this input factor. It is depicted from the graphs that overland flow sensitivity is only significant to changes in M value when assessing the second range of change (-50% to 0%) per the Dry and Wet rainfall scenario, and entirely negligible per all the remaining ranges and also with regards to the Normal scenarios. This ascertains the little connotation of the M to the sensitivity of the overland flow.

#### 7.5.2.2.4. Slope constant (N)

The initial value of the slope constant (N) used in the soil erosion equation (Thornes, 1990) in the DIS model, was determined empirically, taking into account the analysis by Thornes (1983).

In relation to the soil erosion factor, its sensitivity to changes in the N input is plotted quite important per all rainfall scenarios (Figures 7.1-7.3). In this same context, it may be suggested from the charts that the Dry and Normal scenarios present a very similar pattern with regard to the soil erosion sensitivity. These two graphs, illustrates a positive response of the soil erosion sensitivity when the value of the N decreases and a negative response when the N increases. Therefore, there is a direct proportional relationship between both factors. On the contrary, soil erosion sensitivity to changes in the N in the Wet scenario, although significant, it shows a different pattern in comparison to the rest of scenarios, in terms of a negative response in all ranges of change of the input factor. Once more, it seems that the soil erosion sub-model behaviour in the DIS model performs better under dryer that wetter climatic conditions.

As for the rest of SEP, the sensitivity of the soil moisture content output to changes in the N, reveals a complete unresponsiveness, accounting for the irrelevance of this input factor to this particular analysis. To some extent this insensitivity could be justified by the same discussion, arguing that the N input is an intrinsic component of the soil erosion equation, which is not mathematically nor conceptually related to any other sub-model in the DIS template. This same reason seems to be also valid in relation to the sensitivity of the overland flow to changes in the N input factor, although in this case, the output shows a slightly higher sensitivity in some of the ranges of change of the N in the three rainfall scenarios. In spite of this, the overland flow sensitivity to the N factor is considered negligible.

#### 7.5.2.2.5. Vegetation cover (veg)

The vegetative-cover factor describes the impact of vegetation, or lack of, on potential soil erosion. Simply put the more cover you have the better the protection from erosion. Undisturbed natural systems are usually stable, with good vegetation cover characteristics and low erosion rates. As might be expected, interfering with the natural vegetation can seriously affect erosion rates. Deforestation will increase erosion potential by increasing the exposure of the soil to wind and to water. The assessment of the effect of the vegetative cover input factor on the sensitivity of soil erosion, soil-moisture content and overland flow output factor per all the rainfall scenarios of interest, is illustrated in Figures 7.1, 7.2 and 7.3.

With respect to the soil erosion output, the expected sensitivity pattern to changes in the vegetative cover is confirmed in the Dry rainfall scenario but partly observable in the Normal and Wet. Concerning the Dry scenario, the aforementioned assumption of an inverse relationship between vegetation cover and soil erosion is verified. Thus, in the case of a low rainfall amount, it is suggested that when the vegetation cover of the soil decreases, sediment yield experiments a remarkable increase, leading to more important erosion processes in terms of land degradation. The opposite situation is also demonstrated per this scenario (Figure 7.1). Regarding the two remaining rainfall scenarios (Normal and Wet), the previously described pattern is not totally accomplished across all ranges of change of the vegetation cover. On the contrary, the expected inversely proportional relationship between changes in vegetation cover and soil erosion sensitivity is only achieved when addressing changes in the third and fourth ranges of change of the vegetation input factor. Here, in terms of magnitude, the soil erosion sensitivity is very important and is presented as a negative response in the X axis of the graphs, showing a significant decrease of its value. This confirms that the more vegetated the environment, the less prone to suffer soil water or/and wind erosion processes, accounting for the important role of the vegetation cover as a protection factor to the soil. On the other hand, concerning the first two ranges of change of the vegetation cover in the Normal and Wet scenarios, the sensitivity of soil erosion instead of experimenting a positive increase, as it is revealed in the Dry scenario and as it would be expected, its value decreases considerably. Unexpectedly, the sensitivity pattern of soil erosion output in these two scenarios was completely opposed to the Dry one, although in terms of magnitude the sensitivity is shown quite important. However, in absolute terms, the soil-erosion sensitivity to changes in the vegetative cover is remarkable significant per all the assessed ranges of change and rainfall scenarios. It may be drawn from this discussion that the DIS model performs presumably better under drier that wetter climatic conditions.

The sensitivity analysis conducted in order to assess the sensitivity of the soil moisture content to changes in vegetation cover shows a total insensitivity of this output. This unresponsiveness of the soil-moisture content is also verified for the overall set of input parameters (SEP) selected for this testing. In the DIS model, the vegetation cover input is one of the several main factors composing the soil erosion equation and for instance the soil erosion sub-model. Therefore, because of the inexistent direct interrelationship between the soil erosion and the soil-moisture content sub-model, as designed in the DIS template, it is rational from a mathematical point of view the low soil moisture sensitivity to changes in veg. Thus, in terms of magnitude soil moisture output is presented as unresponsive to the vegetation cover, although the showed pattern per all rainfall scenarios is correct. This pattern is conceptually reasonable, ascertaining a direct proportional relationship between the soil moisture and vegetation cover. In this context, it is apparent in the charts which show that when the vegetative cover reduces, then the soil moisture content also decreases, and vice versa. This is quite understandable since the more vegetation covering the soil, the more protection not only to soil erosion processes but also to soil moisture losses through direct soil evaporation. The opposite situation is also plausible. In general, the sensitivity of soil moisture to changes in vegetation cover is considered irrelevant to this analysis.

Taking into account all rainfall scenarios, the whole sensitivity of overland flow output to changes in the vegetative cover is quite low, and specially in the Normal scenario. A dissimilar pattern of the overland flow sensitivity is shown for all rainfall scenarios. Logically what it would be expected is an inverse proportional relationship between the input of change and the output factor, since from the literature is quite apparent that when the vegetation cover of the soil increases, there is a greater interception of the rainfall and consequently the processes of crusting and sealing of the soil reduce considerably, leading to an increase on the infiltration capacity of the soil and for instance reducing the runoff generation. However, as mentioned in the previous section, the vegetation cover factor is only affecting the soil erosion sub-model in a direct way and this is not related mathematically in the DIS model to the overland flow sub-model. As a result, the sensitivity of the overland flow to changes in this specific input factor is not very important in any of the rainfall scenarios.

### 7.5.3. SENSITIVITY RELATED TO HYDROLOGICAL PARAMETERS OF THE DIS

Figures 7.4, 7.5, and 7.6 illustrate the sensitivity of soil moisture content, overland flow and soil erosion to changes in hydrological parameters (HYP) such as: b values, Hortonian constant, soil porosity, psychrometric constant, Priestley and Taylor constant, and saturated hydraulic conductivity. The sensitivity analysis was performed per three rainfall scenarios, Dry, Normal and Wet respectively. These parameters have been grouped as HYP because indirectly or directly they influence one or more hydrological sub-models (SM, PI, GR and PET) in the DIS template.

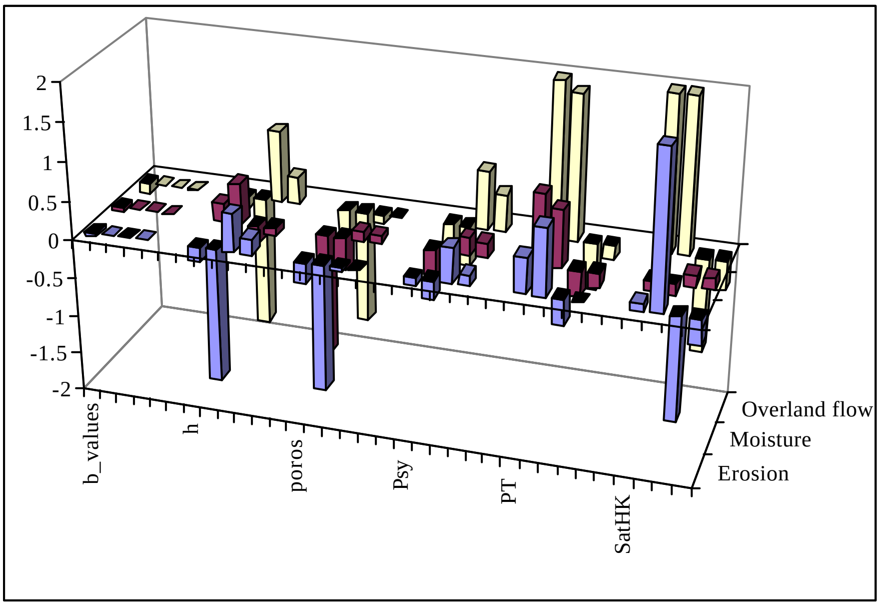


Figure 7.4. Dry rainfall scenario. Sensitivity of soil erosion, soil moisture content, and overland flow to changes in hydrological parameters (HYP)

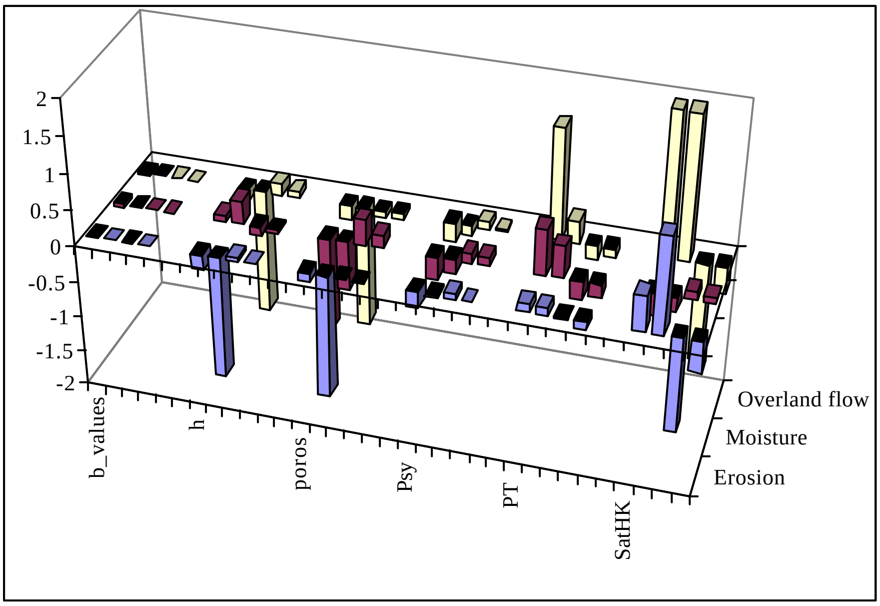


Figure 7.5. Normal rainfall scenario. Sensitivity of soil erosion, soil moisture content, and overland flow to changes in hydrological parameters (HYP)

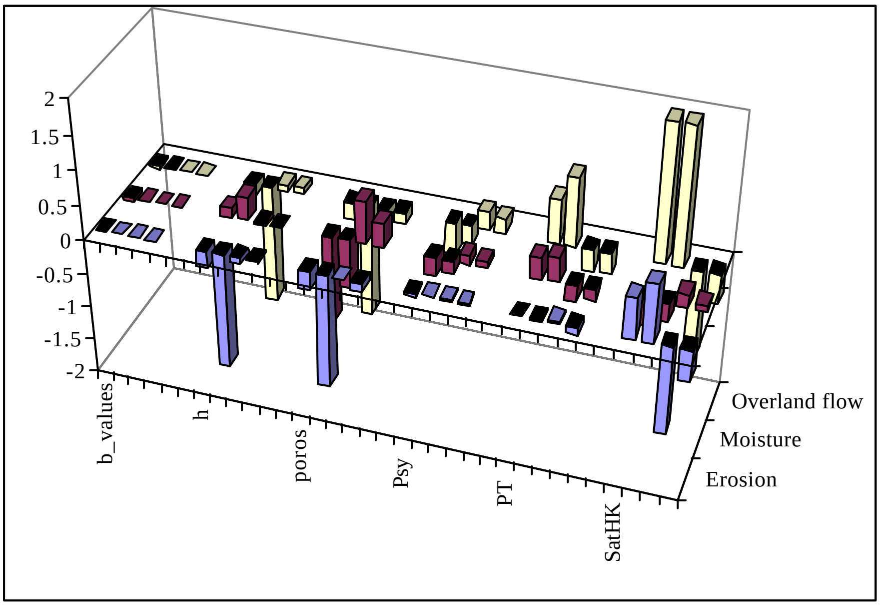


Figure 7.6. Wet rainfall scenario. Sensitivity of soil erosion, soil moisture content, and overland flow to changes in hydrological parameters (HYP)

#### 7.5.3.1. Overall patterns

##### 7.5.3.1.1. Soil moisture content sensitivity to changes in HYP

Soil moisture content (SM) presents a relative high sensitivity to changes in almost all hydrological parameters. This output only shows unresponsiveness to the b input, as presented in Figures 7.4 to 7.6. In this context, considering the overall range of change in the overall set of HYP, the ones related to soil porosity and the Priestley-Taylor constant have the greatest SM sensitivity per all rainfall scenarios. In terms of magnitude, SM also shows sensitivity to saturated hydraulic conductivity, and the Hortonian and psychrometric constant, although it is slightly lower per all scenarios.

#### 7.5.3.1.2. Soil-erosion sensitivity to changes in HYP

Figures 7.4 to 7.6 suggest that the soil-erosion rate (SE) has in general a decreasing sensitivity to changes in HYP, from the Dry, to the Normal and Wet rainfall scenario respectively. Thus, conducting the same sensitivity analysis, less rainfall produces a higher amount the more sensitivity in the SE output. The SE is least responsive to changes in the b factor for all scenarios, making this input completely insensitive. Both, the psychrometric and the Priestley-Taylor constants show a moderate influence on the SE sensitivity in the Dry rainfall scenario, whereas their contribution in the Normal and the Wet scenarios can be considered unimportant in terms of SE sensitivity. Hence, the greatest sensitivity of the SE output is due to changes in the Hortonian constant, the soil porosity and mainly the saturated hydrologic conductivity. In general, the sensitivity of SE to HYP is quite remarkable. This is mainly due to the importance of HYP on the overland flow sensitivity since this output factor directly affects soil erosion as designed in the DIS model and as it happens in reality in the field.

#### 7.5.3.1.3. Overland-flow sensitivity to changes in HYP

As for the two output factors previously analysed, overland flow (OF) sensitivity to changes in HYP shows a slight decrease from the Dry to the Normal and the Wet rainfall scenarios. OF shows unresponsiveness to changes in b parameters, entailing its unimportance to this sensitivity analysis. On the contrary, OF sensitivity is particularly relevant to changes in the remaining HYP. Tables 7.4 to 7.6 suggest that the saturated hydraulic conductivity is the most important and determinant parameter in terms of magnitude, showing a ratio of dI%/dO% higher than +2% in all rainfall scenarios for the two negative ranges of change: -100% to –50% and –50% to 0%. Illustrations show that OF sensitivity follows the same pattern that soil erosion sensitivity taking into consideration all HYP and per all different scenarios, so a strong direct relationship between these two outputs is clear. In fact, because the OF sub-model is extremely influenced by HYP and it is forming part of the soil-erosion component in the DIS model, soil erosion is directly affected by the behaviour of OF output.

#### 7.5.3.2. Hydrological parameters

##### 7.5.3.2.1. B value (b)

The original b input factor (b) is associated to the groundwater recharge sub-model in the DIS template. The b parameter is computed in the Hydropar model (see chapter six) through the expression: b = -2 ?$_{es}$ + ?$_{g}$, where ?$_{es}$ is the air-entry water potential and ?$_{g}$ is the geometric standard deviation of the particle diameter weighed by the fractional proportion of each textural class. Thus, the b factor is mainly based on the textural class of the soils, and therefore it has a specific value per each of the ten selected land uses in the DIS.

It is apparent from the plots associated to the three rainfall scenarios (Fig. 7.4-7.6), that the effect of the b to the sensitivity of the entire set of output factors is completely negligible. Therefore, it might be concluded that neither soil erosion nor soil moisture content or overland flow outputs are significantly sensitive to changes in the b. Accordingly, the b input factor considered totally insensitive regarding the conducted sensitivity analysis, does not play an essential role on the performance of the overall DIS model.

##### 7.5.3.2.2. Hortonian constant (h)

The Horton's equation is used in the DIS model in order to quantify the infiltration process and specifically to determine the potential infiltration capacity of the soils of study (Horton, 1940). This exponential expression of the Hortonian regime includes the Hortonian constant (h) as one of its main components. The h constant is computed empirically on the basis of two parameters, the soil moisture content, computed from fieldwork determinations and the saturated hydraulic conductivity calculated in the Hydropar model. A single and unique h value for all the land uses has been determined from the calculation process that consisted of fitting the two aforementioned parameters to the Horton infiltration equation.

The h constant is an important factor in the potential infiltration sub-model (PI). Thereby, when the value of h decreases, it results in an increase of the PI, and vice versa. From this, interrelationships among several components in the DIS might be explained. To a large extent, it is logical to assume that the increase of the PI value, involves an increase of the actual infiltration capacity (AI), which results in a reduction in the runoff generation and the overland flow (cell based runoff) and subsequently the soil erosion output will also experiments a decrease. In this same way, if the AI increases, the soil-moisture content also increases and since the soil moisture is directly related to the actual evapotranspiration (AET), the latter also increases as also does the groundwater recharge (GR). This sequence of interrelationships among several of the some components in the DIS model is also confirmed when considering the opposite situation.

With reference to the sensitivity of the soil-erosion output to changes in the h constant, it is found to be significant for all rainfall scenarios, since it directly affects infiltration and therefore overland flow and soil erosion (Figures 7.4-7.6). In addition, it seems that the soil-erosion sensitivity follows the same pattern when comparing all the graphs. Here, when the value of the h constant decreases the result is a negative response of the sensitivity of the soil erosion output, which is more important in the second than the first range of change of this specific input. The opposite situation is also confirmed. Thus, when the value of the h constant increases, the sensitivity of the soil erosion exhibits a positive response in the X axis of the charts. Therefore, there seems to be a direct proportional relationship between the h constant and the soil erosion output factor. However, in absolute terms the sensitivity is lower in the third and fourth ranges of change than in the first two. This lower soil erosion sensitivity is much more important when assessing the Normal and Wet scenarios than the Dry one. In fact the responsiveness of the soil erosion to changes in h regarding the two last ranges of change is so little that when assessing them in the Normal and Wet scenarios, the soil-erosion sensitivity can be considered to be irrelevant. The observed pattern could be explained by the aforementioned explanation about the interlinkages among the different components of the DIS model, including the soil-erosion component. In summary, if the h constant decreases then PI increases leading to an increase in the overland flow and consequently the soil erosion process also reduces, and vice versa. This same discussion about the sensitivity of the soil erosion factor to changes in the h is also valid when addressing the sensitivity of the overland-flow output.

Concerning the soil-moisture content sensitivity to changes in the h constant, it is suggested to have the opposite pattern than for soil erosion and overland flow. However, this testing shows a slightly lower responsiveness for all the scenarios, and especially for the Normal and Wet scenarios with regards to the third and fourth ranges of change of the input factor. The identical statement employed in the discussion of the other outputs analysed is applicable to the pattern of the soil moisture sensitivity regarding both, the negative and positive ranges of change of the h constant. Thus, both conceptually and mathematically in the DIS model, when h undergoes a decrease in its value, there is an increase of the potential infiltration capacity as well as of the actual infiltration, which consequently affects the soil moisture of the soil increasing its content. Here, the opposite process is also verified, although it is shown as less important in terms of magnitude of the output sensitivity.

In general it can be said that the three outputs analysed are highly sensitive to the h constant per all assessed rainfall scenarios. Specifically the h constant might be considered an important parameter of the overall DIS model, since it influences directly or indirectly almost all of the sub-models.

#### 7.5.3.2.3. Soil porosity (poros)

In the DIS model, soil porosity (poros) was computed as a function of the bulk density (dry density of the soil) and the particle density (weighted average density of the mineral grains making up the soil) of the soil. This calculation was conducted individually for every soil related to each of the ten selected land uses in the study area. Thus, the porosity of the soil is employed as an indirect measurement of the bulk density. As the porosity of the soil decreases, the available porespace within the soil profile also decreases leading to a reduction of the surface area for drainage from the base of the profile. This decrease potentially produces a higher soil-moisture fraction but there is a corresponding increase in the hydrological flux rates leading to lower soil-moisture fraction. Therefore, as soil porosity decreases, the bulk density increases and the maximum possible soil-moisture content is reduced. The inverse process also occurs.

Figures 7.4 to 7.6 suggest that soil erosion is highly sensitive to reductions in soil porosity particularly in the range of -50% to 0%. However, soil erosion is totally unresponsive in the positive ranges of change of the soil porosity for all rainfall scenarios. The pattern and the magnitude of soil-erosion sensitivity is very similar in all the diagrams, accounting for the insignificant effect of the amount of rainfall on this specific testing. Here, as previously mentioned, the reduction of soil porosity directly affects soil moisture by decreasing its content. Indirectly, soil porosity influences other parameters in the DIS model, through the soil moisture content. In particular, the reduction of soil moisture involves an increase of the potential and actual infiltration capacity of the soil, and the consequent decrease in the runoff generation and overland flow (cell based runoff in the DIS). Certainly, a reduction in runoff generation always implies a decrease in the sediment yield. This result is manifested in a reduction of the soil-erosion rate as suggested in the graphs under discussion. In conclusion, it seems that soil porosity plays an important role on the soil erosion sensitivity when the porosity decreases, whereas soil erosion is insensitive to changes in soil porosity in the ranges of 0 to +50% and +50% to +100%. The exact pattern and magnitude of soil-erosion sensitivity to changes in the soil porosity is shown in the assessment of the overland flow output. Therefore, the same aforementioned arguments are applied to the overland flow when addressing its sensitivity to changes in soil porosity.

The same arguments about the sensitivity of soil erosion to changes in soil porosity can be applied to the soil moisture and overland flow outputs. With reference to the soil-moisture content, in general this output shows significant sensitivity to all ranges of change of the soil porosity. However, the pattern and magnitude of soil-moisture sensitivity is slightly different from that of the soil erosion. By far, the highest sensitivity is found for changes of the soil porosity in the range of -100% to –50%, followed by the –50% to 0% and so on. Here, the sensitivity of soil moisture is important with regards to the positive ranges of change of the input, although the magnitude is lower than in the negative ranges. However, accounting for the magnitude of the overall sensitivity of the soil moisture, it is considered more significant in comparison to the soil erosion and overland flow output factors. The pattern of soil-moisture sensitivity illustrated in the three graphs associated to the rainfall scenarios is justified on the basis of the aforementioned discussion about the interrelationship between soil porosity and soil moisture. In the DIS model, the soil porosity acts as a restricting factor in the soil moisture sub-model. Concretely, this sub-model states that since conceptually, the potential maximum soil-moisture content is equivalent to the maximum available pore space in the soil profile, soil-moisture content can never exceed the soil porosity of the soil. Otherwise, the soil-moisture surplus will lead to runoff generation.

To some extent, the relative insensitivity of all the assessed outputs to changes in the third (0% to +50%) and fourth (+50% to +100%) range of change of the soil porosity, could be explained assuming that in the case of a great increase in the soil porosity, the soil moisture will not increase proportionally. This outcome seems to be realistic since the study area is located in the Mediterranean region where the content of soil-moisture is limited to a large extent by the scarcity of the rainfall characteristics. Therefore, it is suggested that the soil-moisture content largely depends on the rainfall amount.

#### 7.5.3.2.4. Psychrometric constant (Psy)

The psychrometric constant (Psy) is a component of the potential evapotranspiration sub-model (PET) in the DIS template. The Priestley-Taylor equation (Priestley and Taylor, 1972) was selected for describing the PET process in the overall model (see chapter six). This parameter is computed as a function of the specific heat (Cp), the atmospheric pressure (P), the ratio of molecular weight of water vapour to dry air (e) and the latent heat of vaporization (?).

The sensitivity of the three output factors: soil erosion, soil moisture and overland flow, are similar in pattern and magnitude across the whole range of Psy. The output factors, present negative response to the reduction of the input factor and positive response to the increase of the Psy. However, in terms of magnitude, the sensitivity of the soil erosion factor decreases in importance from the Dry to the Wet rainfall scenarios. In particular this output seems insensitive to changes in the Psy forr the Wet rainfall scenario. Therefore, the sensitivity of soil erosion is quite dependent on the rainfall characteristics, being more sensitive when the climate conditions are drier. Regarding the other two outputs, it seems that their magnitude in terms of sensitivity to Psy is similar for all rainfall scenarios (Figures 7.4-7.6). So, the sensitivity of both output factors, the soil moisture content and the overland flow, presumably account for their unresponsiveness to variations in the amount of rainfall.

It is reasonable to assume that the sensitivity pattern of all the mentioned output factors to changes in the Psy, is based on indirect interlinkages among the Psy constant and several components of the DIS model. The Psy constant occurs in the denominator in the Priestley-Taylor PET equation. That means that the Psy influences the PET outcome in an inverse way. Therefore, when the Psy increases, the PET sub-model experiments a decrease in its value. Consequently, if the PET decreases, the actual evapotranspiration (AET) also diminishes. In this same context, the AET component is a subtraction factor in the soil-moisture content equation, which means that if the AET lowers, the soil moisture content increases. In the expression for the calculation of the potential infiltration (PI) under Hortonian conditions, the increase in the soil-moisture content results in a decrease of the PI. Consequently, if the PI decreases the actual infiltration also decreases, and as a result the runoff generation increases and so the overland flow. The last affected factor in this chain of effects is the soil-erosion sub-model, which experiences a substantial increase in its rate when taking into account a higher value of the overland flow. The aforementioned discussion verifies the sensitivity pattern of the soil erosion, soil moisture and overland flow factors to changes in the 0 to +50% and the +50% to +100% ranges of the Psy constant, for all three rainfall scenarios of study. Regarding the –100% to –50% and the –50% to 0% ranges of change of the Psy, the inverse pattern is found.

In general, the remarkable sensitivity of the three output factors to changes in the Psy was slightly unexpected in terms magnitud, since the Psy constant is not considered a primary component in the DIS model but a second order parameter forming part of the PET sub-model. Therefore, the contribution of the Psy in terms of the importance of its value for the overall DIS is apparent and also justified in the discussion of the associated graphs.

#### 7.5.3.2.5. Priestley and Taylor constant (PT)

The Priestley and Taylor empirical constant (PT) is a multiplier factor in the Priestley-Taylor (Priestley and Taylor, 1972) equation of the potential evapotranspiration (PET) sub-model in the DIS template. Therefore, from a mathematical point of view, when the value of PT is increased, the PET also augments, and vice versa, showing a direct relationship between them. Subsequently, if the PET increases, the actual evapotranspiration (AET) outcome also increases. As mentioned in the previous section, the AET operates as a subtraction factor in the soil-moisture expression. As a consequence, when the AET augments the soil moisture content is reduced. In this situation, the higher value of soil moisture entails an augmentation of the potential infiltration (PI) sub-model, and in consequence the value of the actual infiltration (AI) also rises. Accordingly, a decline in the AI implies an increase in the runoff generation and overland flow. As a result, a higher soil-erosion rate is expected from the soil erosion sub-model. Conversely, the aforementioned discussion about the effect of PT changes on the outcomes of several sub-models is also certain. In the context of this analysis, the sensitivity of the soil erosion, soil moisture content and overland flow factors to changes in the PT constant follow the same aforementioned described pattern for all precipitation scenarios (Figures 7.4-7.6). Thus, it is reasonable to assume that when the PT increases, soil erosion, soil-moisture content and overland flow experience a significant increase in terms of sensitivity, and vice versa. The amount of rainfall seems to be relevant to the sensitivity of the outputs, since it is clear from the charts that the magnitude of the sensitivity for the three of them decreases remarkably from the Dry to the Normal and Wet rainfall scenarios. Particularly, the soil-erosion sensitivity can be considered totally irrelevant for the Wet rainfall scenario across all ranges of change of the PT. Therefore, this testing is more significant when addressing lower rather than higher amounts of rainfall. Comparing the three outputs of analyses, the overland flow is the most sensitive to the PT constant, showing an extremely high sensitivity response with regards to the -100% to -50% and –50% to 0% range of change of this input and also per all rainfall scenarios but specially for the Dry one. The soil-moisture content is considered as the second most sensitive output in this study, and the soil erosion the most unresponsive one. Hence, the contribution of the PT constant is notable in the overland-flow sensitivity.

Soil erosion, soil-moisture content and overland flow, show the opposite pattern in terms of sensitivity to changes in the PT constant in comparison to the Psy, for all rainfall scenarios. Comparing the magnitude of the sensitivity of the output factors to changes in the PT and Psy inputs, soil erosion reveals a similar magnitude in both of them, whereas for the soil moisture and overland flow it is higher in relation to the PT rather than the Psy. Therefore, with regard to these two factors of the PET sub-model, the PT constant seems more important in terms of its contribution to the sensitivity analysis, and in particular to the soil moisture and overland flow.

#### 7.5.3.2.6. Saturated hydraulic conductivity (SatHK)

The saturated hydraulic conductivity (SatHK) is a highly variable soil property strongly related to soil structure, and very influenced by effects such as macropores, stones, fissures, cracks and other irregularities formed for various biological and mechanical reasons. SatHK is the factor relating soil water flow rate (flux density) to the hydraulic gradient and is a measure of the ease of water movement through the soil and for instance the reciprocal of the resistance of soil to water movement. In this same context, large, continuous pores have a lower resistance to flow and thus a higher conductivity than small or discontinuous pores. Therefore, soils with high clay content generally have lower hydraulic conductivities than sandy soils because the pore size distribution in sandy soil favours large pores even though sandy soils usually have higher bulk densities and lower total pore space than clayey soils. In the DIS model, the SatHK was individually calculated per each of the ten land uses under study on the basis of the Hydropar hydrological model (see chapter six). Therefore, the SatHK rates are expressed in relation to bulk density and soil texture.

Essentially the SatHK parameter is an important factor in two of the entire set of processes composing the DIS model: the groundwater recharge (GR) and the potential infiltration capacity (PI). Bearing in mind that the study area is characterized by Mediterranean climate conditions, where the incidence of rainfall is subject to strong variations in magnitude and frequency, it is assumed that throughout most of the year soil moisture is always too low for recharge to be a significant component of the soil-water budget. However, recharge could be very high during large rainfall events, since they can produce high soil-moisture contents. To some extent, the SatHK value determines the magnitude of this recharge process, although it is more likely to determine the rapidity of the event than to have a relatively high impact on the total water lost. Therefore, in this discussion the SatHK is considered to have a more significant effect on the potential infiltration capacity of the soil than on the recharge process. In the DIS model, the PI is defined on the basis of a Hortonian regime, and is considered as a complex function of the soil-moisture capacity ($m^3$water/$m^3$soil) and the change in porosity and thus hydraulic conductivity with depth through the soil profile.

As might be expected, the most sensitive output to SatHK across all of its range of change is the overland flow, accounting for the high SatHK impact on the hydrological fluxes in the DIS model. A similar pattern of the overland sensitivity is found in all rainfall scenarios (Figures 7.4-7.6). Here the overland-flow factor seems to be extremely sensitive to changes in the negative ranges of the SatHK (first and second range of change) and slightly less to the positive ones, although in general the sensitivity of this output is obviously highly significant to the SatHK. The magnitude of the sensitivity is the same for all the precipitation scenarios accounting for the low sensitivity of the amount of rainfall in this testing, when the rest of parameters remain invariable.

The second most sensitive output to changes in SatHK is the soil erosion. Although it can be considered to be quite sensitive across all the values of SatHK, it is slightly different in pattern and magnitude to the overland-flow sensitivity with regards to the negative ranges of this input. The magnitude of the soil-erosion sensitivity in the -100% to –50% range is quite low, especially in the Dry rainfall scenario, whereas the overland-flow sensitivity is extremely high. Regarding the positive ranges of change of the SatHK, both output factors seem equally sensitive in terms of pattern and magnitude.

The soil-moisture content factor is found to be the most unresponsive one with regard to the soil erosion and overland flow. In addition, soil-moisture content has the opposite pattern and is much lower in magnitude than the other two studied outputs for all rainfall scenarios.

The sensitivity pattern of the three analysed output factors concerning changes in the SatHK will be discussed on the basis of the aforementioned assumption, which states that SatHK mainly affects the potential infiltration capacity component in the DIS template. Therefore, an increase in the SatHK factor results in an increase in the potential infiltration capacity of the soil (PI), and vice versa. In this case, when PI increases, the actual infiltration capacity (AI) also increases, and consequently the overland flow will be lower. As a result, soil-erosion rates are also lower, since the overland flow has a direct influence on this process. Conversely, the opposite sensitivity pattern of the soil-moisture content, is justified through the effect of the SatHK on the groundwater recharge component (GR). Here, the SatHK is also a main factor of the GR expression and it is directly related to the GR outcome, since when the SatHK increases, it is clear that the water recharge of the soil profile also increases. Therefore, the SatHK has an indirect effect on the soil-moisture content, through the relationship between the GR and this output factor in the soil moisture content sub-model. The lower sensitivity of the soil moisture accounts for a less relevant role of the SatHK on the GR sub-model. The augmentation of the GR value, involves a decrease in the soil moisture output, since it is a subtraction factor in the equation associated to the soil moisture sub-model. In addition, a reduction in the soil moisture content will contribute to a higher potential infiltration capacity. Thus, the SatHK directly and indirectly influences the value of PI and all the sub-models using these parameters. So, it is apparent all the chain of effects across the several sub-models in the DIS template, due to changes in the SatHK.

## 7.6. KEY PARAMETERS FOR SOIL EROSION IN THE DIS MODEL

Table 7.1 is a summary of the sensitivity of the three main outputs of the DIS model - the soil erosion, soil moisture content and overland flow sub-model - to changes in the overall set of soil-erosion related parameters (SEP): soil depth (depth), soil erodability factor (K), discharge scaling factor (M), slope constant (N) and vegetation cover (veg). The table illustrates per each of the three rainfall scenarios of study (Dry, Normal and Wet), the range of change associated to the maximum sensitivity and insensitivity of each SEP.

Table 7.1. Maximum (+ or -) sensitivity and insensitivity of the output factors to changes in soil erosion related parameters (SEP) per each rainfall scenario

| **Maximum (+ or -) sensitivity and insensitivity factor (dO%/dI%)<br>se: maximum sensitivity; in: maximum insensitivity** | | | | | | |
|---|---|---|---|---|---|---|
| **R. Scenarios** | | **Depth** | **K** | **M** | **N** | **Veg** |
| | | | | **Soil moisture content ($m^3/m^3$)** | | |
| **Dry** | se | 0.0805 (b) | 0.0449 (a) | -0.2279 (c) | -0.0114 (a) | -0.0106 (a) |
| | in | 0.0185 (a) | -0.0065 (d) | 0.0002 (a) | 0.0016 (d) | $6.8x\ 10^{-5}$ (d) |
| **Normal** | se | -0.1364 (a) | 0.0129 (b) | -0.0636 (c) | -0.0249 (a) | -0.0457 (b) |
| | in | 0.0034 (d) | -0.0065 (d) | 0.0013 (a) | 0.0139 (d) | 0.0161 (d) |
| **Wet** | se | -0.2323 (a) | 0.1163 (a) | 0.1433 (b) | -0.0549 (b) | -0.0856 (b) |
| | in | 0.0466 (d) | -0.0115 (d) | 0.0039 (a) | 0.0347 (d) | 0.0411 (d) |
| | | | | **Overland flow (mm/month)** | | |
| **Dry** | se | 0.3997 (b) | 0.5827 (a) | 0.6439 (b) | 0.1963 (c) | 0.1993 (d) |
| | in | 0.0249 (a) | -0.0178 (d) | 0.0155 (a) | -0.0472 (a) | 0.0049 (a) |
| **Normal** | se | -0.2037 (c) | -0.0273 (b) | 0.1364 (c) | 0.0619 (b) | 0.0861 (b) |
| | in | 0.0984 (a) | -0.0032 (a) | 0.0119 (b) | -0.0097 (d) | -0.0179(d) |
| **Wet** | se | 0.1936 (a) | 0.7713 (a) | 0.7346 (b) | 0.1824 (d) | 0.2007 (d) |
| | in | 0.0002 (c) | 0.0092 (c) | 0.0485 (a) | 0.0257 (b) | 0.1032 (c) |
| | | | | **Soil erosion (mm/month)** | | |
| **Dry** | se | -1.4476 (a) | -1.6476 (a) | -1.8035 (b) | -0.5817 (c) | -0.9460 (c) |
| | in | 0.0816 (d) | 0.1460 (d) | -0.1299 (c) | 0.2648 (a) | -0.5457 (d) |
| **Normal** | se | -1.0971 (a) | -1.7076 (a) | -1.7932 (b) | -0.5047 (c) | -0.8636 (a) |
| | in | 0.5336 (d) | 0.0384 (d) | -0.2002 (a) | 0.1776 (a) | -0.80870 (b) |
| **Wet** | se | 1.0937 (c) | -1.9906 (a) | -1.5868 (b) | -0.4855 (a) | -0.8736 (a) |
| | in | -0.9569 (a) | 0.0094 (b) | -0.4066 (a) | -0.1393 (b) | -0.4727(d) |

Depth: soil depth (mm); K: erodability constant (dimensionless); M: discharge scaling factor (dimensionless); N: slope constant (dimensionless); Veg: vegetation cover (%); se: maximum sensitivity value; in: maximum insensitivity value. In brackets: (a) corresponds to the range of input change from –100 to –50%; (b) from –50 to 0%; (c) from 0 to +50%; and (d) from +50 to +100%.

As it has been mentioned in the discussion of the soil-moisture sensitivity to every single SEP, these parameters show very low response of the entire set of rainfall scenarios. The maximum soil moisture sensitivity values to all SEP, ranges from –0.086 (veg) to –0.232 (depth), and they are mainly related to the negative ranges of change of the input factors and principally concentrated in the Wet rainfall scenario. Therefore, as presented in Table 7.1, with regards to the three rainfall scenarios, it seems that the most important sensitivity is associated to the highest rainfall scenario, although in terms of magnitude it is considered negligible. Concerning the maximum soil moisture insensitivity values, they range from –0.0065 (K) to +6.8x10$^{-5}$ (veg), and they are mostly associated to changes in the range +50% to +100%, and essentially concentrated in the Dry rainfall scenario. Hence, as illustrated in Table 7.1, the most unresponsive results are found in the Dry scenario. Thereby, it seems clear from the results that the highest soil-moisture sensitivity is related to the Wet rainfall scenario and the lowest to the Dry one, leading to the conclusion that the rainfall amount plays a relatively significant role in this particular analysis. In relation to the most and least relevant parameters on the soil-moisture sensitivity testing, these are the ones presenting the highest and lowest response in terms of magnitude, respectively. Thus, the most relevant parameter is the soil depth and the most insensitive is the one related to the vegetation cover.

With reference to the overland flow, the maximum sensitivity ranges from +0.1963 (N) to +0.7713 (K), and the minimum from –0.0097 (N) to +0.0002 (soil depth). The maximum and minimum sensitivity values are mostly concentrated in the Wet and Normal rainfall scenarios respectively. In particular the N and veg parameters show a similar response in terms of magnitude of the maximum overland flow sensitivity in both, the Dry and Wet rainfall scenarios. Moreover, results in Table 7.1 do not show a specific pattern in the range of change of the SEP regarding their maximum and minimum sensitivity values. Therefore, the sensitivity of this output seems not to be dependent on the rainfall amount and seems quite irregular across the whole ranges of change of the SEP. Table 7.1 suggest that the most relevant parameters for the sensitivity analysis of this output are the soil erodibility factor and the discharge scaling factor. However, it is not clear from the results which is the most irrelevant parameter in this testing. In conclusion, it can be said that the highest sensitivity is related to the scenario with the highest rainfall amount (Wet). However, the role of the rainfall amount concerning this particular output is not apparent.

The maximum soil erosion sensitivity values range from –0.5817 (N) to –1.9906 (K), and the minimum from –0.4727 (veg) to +0.0094 (K). The maximum sensitivity results are mainly concentrated in the Dry rainfall scenario, although there is very little difference with the Normal and Wet scenarios, accounting for a relative small significance of the rainfall amount to this testing. On the other hand, the minimum sensitivity results are mostly related to the Wet rainfall scenario. Here, soil depth, soil erodibility, and discharge scaling factor are presented as the most significant parameters for the soil erosion sensitivity in terms of magnitude. The other two parameters, the N and veg, although being relevant, are less important in magnitude. From Table 7.1 it seems that the K is the most remarkable input above all the rest. Regarding the ranges of change of the maximum and minimum sensitivity values, there is no a clear pattern relating the assessed SEP.

It is obvious from the results that comparing the three output sensitivities to the entire set of SEP, the soil erosion is by far the most sensitive and the soil- moisture content the least sensitive output.

## 7.7. KEY PARAMETERS FOR HYDROLOGY IN THE DIS MODEL

Table 7.2 is a summary of the sensitivity of the three main outputs of the DIS model - the soil erosion, soil-moisture content and overland-flow sub-model - to changes in the overall set of hydrological related parameters (HYP): b, Hortonian constant (h), soil porosity (poros), psychrometric constant (Psy), the Priestley and Taylor constant (PT), and the saturated hydraulic conductivity (SatHK). The table illustrates per each of the three rainfall scenarios of study (Dry, Normal and Wet), the range of change associated to the maximum sensitivity and insensitivity of each HYP.

The maximum soil-moisture sensitivity values range from –0.0589 (b) to –1.5778 (poros) and they are essentially concentrated in the Dry rainfall scenario. According to Table 7.2, this sensitivity response is associated to the negative ranges of change of the

HYP. On the other hand, the minimum soil-moisture sensitivity values range from –0.1589 (PT) to –4.6x10$^{-6}$ (b), and they are mostly related to the positive ranges of change of the HYP and mainly concentrated in the Wet scenario.

Table 7.2. Maximum (+ or -) sensitivity and insensitivity of the output factors to changes in the hydrological related variables (HYP) per each rainfall scenario

| **Maximum (+ or -) sensitivity and insensitivity factor (dO%/dI%)**<br>**se: maximum sensitivity; in: maximum insensitivity** | | | | | | | |
|---|---|---|---|---|---|---|---|
| **R. Scenarios** | | **b** | **h** | **poros** | **Psy** | **PT** | **SatHK** |
| **Soil moisture content (m$^3$/m$^3$)** | | | | | | | |
| **Dry** | se | -0.0528 (a) | 0.5206 (b) | -1.5778 (a) | -0.3308 (a) | 0.9125 (a) | -0.1446 (b) |
| | in | -1.2x10$^{-6}$ (d) | -0.0938 (d) | 0.0994 (d) | 0.1975 (d) | -0.1885 (d) | -0.1307 (a) |
| **Normal** | se | -0.0431 (a) | 0.3362 (b) | -1.3191 (a) | -0.3107 (a) | 0.6308 (a) | -0.2951(a) |
| | in | -1.2 x10$^{-6}$ (d) | -0.0473 (d) | 0.1702 (d) | 0.1127 (d) | -0.1870 (d) | 0.0785 (d) |
| **Wet** | se | -0.0589 (a) | 0.3470 (b) | -1.2773 (a) | -0.2619 (a) | 0.3693 (b) | -0.3614 (a) |
| | in | -4.6 x10$^{-6}$ (d) | -0.0158 (d) | 0.3506 (d) | 0.1011 (d) | -0.1589 (d) | 0.0947 (d) |
| **Overland flow (mm/month)** | | | | | | | |
| **Dry** | se | -0.1347 (a) | -1.6891 (b) | -1.4678 (b) | 0.7476 (c) | 3.1798 (a) | 8.4580 (a) |
| | in | 7.4 x10$^{-5}$ (b) | -0.1364 (a) | -0.0032 (d) | -0.3874 (a) | -0.1836 (d) | -0.3769 (d) |
| **Normal** | se | -0.0399 (a) | -1.7584 (b) | -1.7032 (b) | -0.2541 (a) | 1.5541 (a) | 5.9075 (a) |
| | in | -2.3 x10$^{-5}$ (d) | 0.0944 (d) | -0.0689 (d) | 0.0265 (d) | -0.1753 (c) | -0.3727 (d) |
| **Wet** | se | -0.0425 (a) | -1.7490 (b) | -1.6596 (b) | -0.4690 (a) | 0.9989 (b) | 4.0046 (b) |
| | in | -0.0001 (d) | 0.0890 (c) | -0.1445 (c) | 0.2161 (d) | -0.3084 (c) | -0.4263 (d) |
| **Soil erosion (mm/month)** | | | | | | | |
| **Dry** | se | -0.0457 (a) | -1.7306 (b) | -1.6411 (b) | 0.4488 (c) | 0.8647 (b) | 2.4501 (b) |
| | in | -6.3 x10$^{-5}$ (d) | -0.1828 (a) | -0.0084 (d) | -0.1012 (a) | -0.0135 (d) | 0.1001 (a) |
| **Normal** | se | -0.0169 (a) | -1.7305 (b) | -1.7410 (b) | -0.2049 (a) | -0.1183 (d) | -1.3184 (c) |
| | in | -9.4x10$^{-6}$ (b) | 0.0136 (d) | -0.0039 (c) | 0.0089 (d) | -0.0221 (c) | -0.4251 (d) |
| **Wet** | se | -0.0137 (a) | -1.6868 (b) | -1.6753 (b) | 0.0179 (b) | -0.0995 (d) | -1.2742 (c) |
| | in | 1.1x10$^{-6}$ (c) | -0.0147 (d) | -5.4 x10$^{-5}$ (c) | 0.0127 (c) | -0.0151 (a) | -0.4363 (d) |

B: b (dimensionless); h: Hortonian constant (dimensionless); poros: soil porosity (m$^3$/m$^3$); Psy: psychrometric constant (dimensionless); PT: Priestley-Taylor constant (dimensionless); SatHK: saturated hydraulic conductivity (mm/hour); se: maximum sensitivity value; in: maximum insensitivity value. In brackets: (a) corresponds to the range of input change from –100 to –50%; (b) to –50 to 0%; (c) to 0 to +50%; and (d) to +50 to +100%.

So, there is a clear pattern in the range of change of the SEP associated to the sensitivity and insensitivity of soil-moisture content. Irrespective of this outcome, the highest sensitivity occurs in the Dry scenario, and the highest insensitivity in the Wet one, accounting for a relevant role of the rainfall amount. The porosity of the soil is presented as the most significant parameter in terms of soil-moisture sensitivity, and the b as the most unresponsive.

With reference to the sensitivity of the overland flow, the maximum values range from –0.1347 (b) to +8.4580 (SatHK), and the minimum from –0.3727 (SatHK) to $-2.3 \times 10^{-5}$ (b). Table 7.2 suggests that overland flow is extremely sensitive to changes in the SatHK in all rainfall scenarios, but in particular in the Dry scenario. Specifically, SatHK values are always above +2 (IP%/OP%), the maximum value of the sensitivity factor considered in this analysis. Therefore the saturated hydraulic conductivity is by far the most important input factor in the overland flow sensitivity analysis. The maximum sensitivity results are essentially related to the negative ranges of the SEP and mainly concentrated in the Dry scenario. On the contrary, the maximum insensitivity values are from the positive ranges of the SEP and mostly associated to the Normal rainfall scenario. Here, the most insensitive parameter is the b, showing a very low response in all the rainfall scenarios.

The maximum soil-erosion sensitivity and insensitivity results range from -0.0457 (b) to +2.4501 (SatHK), and from +0.1001 (SatHK) to $+1.1 \times 10^{-6}$ (b), respectively. As for the overland flow, the maximum sensitivity seems to be associated to the negative ranges of change of the HYP and mainly related to the Dry rainfall scenario. Regarding the maximum insensitivity values, these results seem to be more related to the positive ranges of change of the HYP. However, it can be considered that there is not a clear pattern in the range of change or in the associated rainfall scenario. Here, the strong relationship between overland flow and soil erosion when assessing hydrological parameters is obvious. In this way, parameters linked to the maximum and minimum sensitivity of the overland flow are the same as for the soil-erosion output. Thus, the SatHK and the b are the most and less significant parameters, although in terms of magnitude, both of them show lower values than when assessing overland flow sensitivity. In relation to the sensitivity to changes in the remaining HYP, h and poros are equally important in magnitude (around –1.7) in the three rainfall scenarios. Likewise, the Psy and specially the PT constant, are also important in the soil-erosion sensitivity testing, although much lower in magnitude (+0.4 and +0.8 respectively). On the other hand, these two HYP seems to be only sensitive when addressing the Dry rainfall scenario, and highly insensitive concerning the rest of scenarios.

The maximum sensitivity associated with the outputs evaluated seems to be related to the Dry rainfall scenario (Table 7.2), whereas there is not a clear pattern in the minimum sensitivity values. It may be drawn from Table 7.2 that the model performs better under drier than wetter conditions, since all the three outputs show their highest sensitivity response to changes in the entire set of HYP in the Dry rainfall scenario.

## 7.8. OUTLINE OF THE SENSITIVITY ANALYSIS OF THE DIS

The important role of the rainfall amount to the model sensitivity is more significant when analysing the HYP parameters than the SEP ones. However it is quite evident that in the case of the soil-erosion sensitivity, which is believed the most important output in terms of responsiveness to changes in SEP, the highest sensitivity values are concentrated in the Dry rainfall scenario and the lowest in the Wet. On the other hand with regards to the three outputs sensitivity to HYP, the maximum sensitivity is concentrated in the Dry rainfall scenario, although there is not a clear pattern in the maximum insensitivity response. In conclusion, it is reasonable to assume that the DIS model is significantly sensitive to rainfall amount. Taking into account the fact that the DIS model was specifically developed for Mediterranean conditions and for instance the scarcity of precipitation is an important feature most of the year, to some extent, this template seems to be more sensitive to the rainfall amount when the conditions are drier than wetter as it was previously expected.

As previously discussed in this chapter, due to the non-linear pattern of the DIS model, is quite difficult to rank the sensitivity of the input parameters. However, it is clear the relatively significance of the SEP and HYP parameters to the overall sensitivity analysis. Therefore, the most likely order in terms of the inputs relevance to the overall sensitivity is ascertained. The soil erodability factor is revealed as the key input factor of the SEP, although the discharge scaling factor as well as the depth of the soil are also seem to be extremely important in the sensitivity analysis of the DIS model with regards to soil erosion, which is the most sensitive output. On the other hand, the most unresponsive parameters are the vegetation cover and the slope constant. Regarding the testing to HYP, the most remarkable input is the saturated hydraulic conductivity per all outputs and scenarios of precipitation. Some other significant factors in the analysis are in order of magnitude and for instance importance: the soil porosity, the Hortonian constant, the Priestley-Taylor constant and the Psychrometric constant. The b input is revealed as the most insensitive factor.

Taking into consideration the overall DIS model sensitivity, in terms of magnitude of the sensitivity response, the saturated hydraulic conductivity has been revealed as the most important input. In this same context, the most irrelevant parameter is set as the b, which has shown completely unresponsiveness to the entire set of outputs and rainfall scenarios in the study. Respective of this, accurate and precise data will be required for the SatHK and also for the aforementioned relevant inputs, whereas, data related to the b is not relevant in the DIS model. Both parameters are from the HYP, which might account for the higher relevance of the hydrological than the soil erosion related parameters in the DIS model, as illustrated in the Figures 7.1 to 7.6. In conclusion, it may also be drawn that the DIS model is highly responsive to the Dry rainfall scenario.

# 8. MODEL SIMULATIONS



## 8.1. VALIDATION OF THE DIS MODEL

The accuracy of the Desertification Indicator System (DIS) simulation for a small Mediterranean catchment has been determined by comparing the performance predicted by the model with data collected during fieldwork campaigns. In essence, the validation of the DIS model should allow the assessment of the reliability of the model outcomes and indicate which parts of the model require further development whilst providing an indication of the uncertainty associated with model predictions.

### 8.1.1. MODEL VALIDATION: A BRIEF INTRODUCTION

The process of validating a model is a necessary step in the development of any kind of model. In general, the validation of a model consists of finding out how well a model performs comparing its simulated results with those from the real world (Hardisty et al., 1993). In an ideal scenario, the model validation would be achieved by choosing the model and the external inputs and so that the difference between the validation output and the true system output was identically zero. However, the whole process clearly depends critically upon the quality of the real world data (observed data), which are being used and data quality and appropriateness is an important aspect of model analysis (Hardisty et al., 1993). Thus, the reliability of a model validation is always constrained by the quality and quantity of validation data available. Moreover, it is important to recognize that uncertainty is inevitable in any model since a model, by definition, will never be as complex as the reality it portrays.

## 8.1.2. MODEL ADJUSTMENTS FOR VALIDATION RUNS

### 8.1.2.1. Spatial scale constraints

The DIS model was originally developed for the catchment scale, and in particular for the Serra de Rodes catchment (30 $km^2$). Consequently, the verification and the sensitivity analysis of the model were also carried out for this particular catchment area at the scale relevant for the model use. In theory, this scale should be used also for the validation process. However, the only available fieldwork data for validation purposes are at the plot scale, obtained from the monitoring of the runoff-erosion microplots, one $m^2$ area. Hence, since both the simulated and the observed results must be comparable, both have to be related to the same spatial scale. Therefore, the model validation instead of being performed at the scale at which the DIS was designed for, it is conducted at a one-cell resolution scale (five $m^2$), the smallest possible area at which the model can be run and an area compatible with the microplot data.

Furthermore, since the experimental microplots were of a bounded design, soil from upward the catchment was not able to contribute on the regeneration of the microplot soil fine fraction (clay and silt), which are the most easily mobilized by runoff water and therefore the ones contributing most to the sediment yield. On the contrary, soil experienced a progressive depletion of its fine fraction after each rainfall event. The microplot design not only constrained the validity of the microplot life to a one-year period, but it only limited the runoff and soil erosion yields, to the area of the microplot. This constraint justifies the irrelevance of running the model at the exact geographical location of each microplot in the Serra de Rodes catchment. Results will only depend on the specific characteristics of the microplot. In the validation, fieldwork data were grouped and reorganized according to the several land uses and plot characteristics under study, and they were directly and individually set up in the script for each particular test. So the model was parameterised on the basis of the plot data and not the GIS data.

#### 8.1.2.2. Temporal scale constraints

It is also necessary to take into account the temporal scale in the validation of the DIS template. Both the simulated results and the observed data must be over the same timescale in order to be comparable. Due to the availability of monthly meteorological data, the timestep of the DIS was set up as monthly, entailing that all the mathematical equations in the model were also selected and/or determined accordingly. Thus, another scale problem is revealed here, since microplot field data (i.e. runoff, sediment yield, rainfall amount) were determined on a rainfall event basis for all events instead of being monthly as required in the model. In order to overcome this problem, the amounts of rainfall, runoff and sediment field data from each rainfall event throughout the one-year field observation period have been grouped according to each month of the year, obtaining monthly observed data for each land use under study.

### 8.1.3. DIS MODEL-VALIDATION PROCEDURE

The validation of the model has been performed for each of the representative selected land uses in the catchment, with the exception of the land use set up as “Roads”. The DIS model validation procedure is detailed in appendix 5.

### 8.1.4. DIS MODEL-VALIDATION DATA REQUIREMENTS

Field data can be separated into input parameters and validation data.

### 8.1.4.1. Input field data

The overall set of input maps required to run the DIS was recalculated for each of the nine different land uses at single cell spatial scale (see section 6.3.7). Data were associated to the specific year of field observation (August 1999 - June 2000).

Table 8.1. Input field data per land use for the DIS model validation

| Land use | VC | b | DEM | Moist | Poros | SatHK | Depth | Slope |
|---|---|---|---|---|---|---|---|---|
| **DCT** | 79 | 2.126 | 123 | 0.152 | 0.543 | 9.93 | 290 | 23 |
| **SCT** | 30 | 1.844 | 225 | 0.111 | 0.444 | 7.21 | 150 | 24 |
| **DS** | 80 | 1.886 | 185 | 0.148 | 0.448 | 6.228 | 390 | 18 |
| **SS** | 50 | 1.692 | 200 | 0.112 | 0.411 | 4.568 | 200 | 18 |
| **COT** | 20 | 2.088 | 95 | 0.116 | 0.414 | 3.83 | 400 | 25 |
| **AOT** | 69 | 1.896 | 155 | 0.114 | 0.476 | 5.598 | 250 | 20 |
| **CV** | 14 | 1.791 | 55 | 0.115 | 0.306 | 3.432 | 150 | 18 |
| **PTA** | 78 | 2.231 | 235 | 0.084 | 0.541 | 10.479 | 100 | 25 |
| **BS** | 40 | 1.836 | 165 | 0.123 | 0.421 | 4.009 | 300 | 20 |

VC: vegetation cover (%); b (dimensionless); DEM: digital elevation model (m); Moist: soil moisture content (mm water/mm soil); poros: soil porosity (mm porespace/ mm soil); SatHK: saturated hydraulic conductivity (mm/hr); depth: soil depth (m), slope: slope angle (degrees).

Table 8.2. Meteorological data required for the validation process

| Months | Rainfall (mm) | Temperature (ºC) | Solar radiation (MJ/m2) |
|---|---|---|---|
| **August** | 15.9 | 24.6 | 595.2 |
| **September** | 118.6 | 21.5 | 468.0 |
| **October** | 44.9 | 16.4 | 319.3 |
| **November** | 58.3 | 10.9 | 222.0 |
| **December** | 4.5 | 8.6 | 182.9 |
| **January** | 50.3 | 7.8 | 204.6 |
| **February** | 9.0 | 11.5 | 302.4 |
| **March** | 46.1 | 12.6 | 458.8 |
| **April** | 19.7 | 13.3 | 519.1 |
| **May** | 75.3 | 19.0 | 564.2 |
| **June** | 24.6 | 22.6 | 624.0 |
| **July** | 15.4 | 23.7 | 688.2 |

These maps were related to the variables: vegetation cover percentage (VC), b, digital elevation model (DEM), soil moisture content (Moist), soil porosity (poros), saturated hydraulic conductivity (SatHK), soil depth (depth), and slope angle (slope) (Table 8.1).

Likewise, new input scalar and nominal land-use maps were also computed. Moreover, for operability reasons, the boolean and scalar clone maps were created for one cell and used as constants in the script of every land use validation test.

Time-series files of monthly rainfall and temperature, were created on the basis of field and automatic weather station (AWS) data, respectively (Table 8.2). New monthly solar radiation maps for the year of study were also developed on the basis of one cell using data from the AWS (Table 8.2). They were used in each land-use validation test since the slope aspect effect does not have a strong impact on runoff and erosion.

#### 8.1.4.2. Output field data

Validation data were aggregated across the microplots in each land use and integrated up to monthly averages for comparability with the model outputs.

Table 8.3. Sediment yield per land use and month throughout the study period

| | | Soil erosion per land use (g/m$^2$) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Year** | **Month** | **DCT** | **SCT** | **DS** | **SS** | **COT** | **AOT** | **CV** | **PTA** | **BS** |
| **1999** | **August** | 0.07 | 6.49 | 1.39 | 3.44 | 31.39 | 0.83 | 7.61 | 1.84 | 12.83 |
| | **September** | 2.43 | 33.57 | 17.64 | 17.18 | 765.97 | 17.20 | 51.82 | 110.54 | 101.22 |
| | **October** | 0.51 | 2.13 | 2.60 | 6.87 | 142.56 | 0.97 | 13.09 | 16.91 | 14.90 |
| | **November** | 3.06 | 17.89 | 0.49 | 17.52 | 132.81 | 0.49 | 10.18 | 8.99 | 5.13 |
| | **December** | 0.00 | 1.07 | 0.81 | 0.00 | 20.34 | 0.00 | 13.09 | 8.09 | 0.00 |
| **2000** | **January** | 2.14 | 1.99 | 1.19 | 5.22 | 22.37 | 1.49 | 8.32 | 5.94 | 0.81 |
| | **February** | 0.00 | 0.00 | 0.00 | 4.16 | 50.97 | 0.00 | 0.00 | 0.00 | 0.00 |
| | **March** | 0.57 | 0.12 | 0.11 | 0.63 | 53.88 | 0.00 | 4.91 | 2.77 | 0.84 |
| | **April** | 1.67 | 0.61 | 0.02 | 5.83 | 84.44 | 0.60 | 3.48 | 2.63 | 0.38 |
| | **May** | 1.68 | 3.55 | 0.00 | 8.93 | 31.54 | 0.00 | 1.29 | 4.10 | 0.39 |
| | **June** | 1.08 | 12.31 | 1.21 | 11.03 | 77.20 | 4.99 | 3.49 | 14.17 | 0.97 |
| | **July** | - | - | - | - | - | - | - | - | - |
| Total average values | | *1.20* | *7.25* | *2.32* | *7.35* | *128.50* | *2.41* | *10.66* | *16.00* | *12.50* |

DCT: dense cork trees; SCT: sparse cork trees; DS: dense shrub; SS: sparse shrub; COT: cultivated olive trees; AOT: abandoned olive trees; CV: cultivated vineyard; PTA: pine trees afforestation; BS: bare soil.

Table 8.4. Runoff generation per land use and month throughout the study period

| | | Overland flow (mm/month) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Year** | **Month** | **DCT** | **SCT** | **DS** | **SS** | **COT** | **AOT** | **CV** | **PTA** | **BS** |
| **1999** | **August** | 0.06 | 0.19 | 0.01 | 0.67 | 1.18 | 0.07 | 1.09 | 0.06 | 0.89 |
| | **September** | 4.21 | 21.53 | 3.02 | 17.20 | 20.25 | 6.37 | 10.26 | 10.58 | 5.55 |
| | **October** | 0.15 | 3.97 | 0.84 | 4.26 | 7.77 | 2.49 | 4.38 | 3.29 | 5.35 |
| | **November** | 0.81 | 11.79 | 1.30 | 6.69 | 8.35 | 2.90 | 5.01 | 5.22 | 6.60 |
| | **December** | 0.00 | 0.10 | 0.01 | 0.00 | 0.10 | 0.00 | 0.00 | 0.00 | 0.00 |
| **2000** | **January** | 0.00 | 0.87 | 0.00 | 0.92 | 10.88 | 0.00 | 3.31 | 2.37 | 0.85 |
| | **February** | 0.00 | 0.00 | 0.00 | 0.11 | 0.84 | 0.00 | 0.00 | 0.00 | 0.00 |
| | **March** | 0.00 | 0.90 | 0.03 | 0.99 | 1.72 | 0.28 | 1.71 | 0.99 | 0.19 |
| | **April** | 0.15 | 0.28 | 0.04 | 2.98 | 12.24 | 0.63 | 4.92 | 2.96 | 0.46 |
| | **May** | 0.55 | 0.58 | 0.03 | 3.10 | 4.02 | 0.53 | 2.50 | 1.66 | 0.15 |
| | **June** | 0.24 | 0.53 | 0.21 | 4.79 | 6.55 | 0.21 | 3.38 | 7.45 | 0.14 |
| | **July** | - | - | - | - | - | - | - | - | - |
| Total average values | | *0.56* | *3.70* | *0.50* | *3.79* | *6.72* | *1.23* | *3.32* | *3.14* | *1.83* |

DCT: dense cork trees; SCT: sparse cork trees; DS: dense shrub; SS: sparse shrub; COT: cultivated olive trees; AOT: abandoned olive trees; CV: cultivated vineyard; PTA: pine trees afforestation; BS: bare soil.

These monthly field results for land use are the ones to be compared with the ones simulated by the model. The month of July (2000) was excluded from the analysis since no field data were available for this month.

### 8.1.5. TIMESCALE FOR THE DIS MODEL VALIDATION

The model was run independently nine times, one per each land use to be tested. Although, the validation analysis is based on one-year field observations (August 1999-July 2000), the timestep of the model, which is monthly based, was set up as 24 (2 years), in order to allow the model stabilize its results during the first run (12 timesteps) and this produce more realistic initial conditions for the model test data. Then, the validation assessment will be focussed on the last 12 timesteps, expected to be at the equilibrium state, and therefore, more accurate and reliable.

### 8.1.6. DIS MODEL SIMULATED DATA

The validation of the model uses two of the most important indicators of land degradation: soil erosion and overland flow. Both simulated and observed data must be in the same units in order to be analysed together. Since simulated soil erosion rates are in mm instead of g/m$^2$ as for the observed data, the field bulk density data in g/cm$^3$ were required for their conversion into g/m$^2$ (Table 8.5 – 8.7.). Overland flow simulated and observed data are in the same units and therefore no calculation is needed.

#### 8.1.6.1. Soil erosion simulated data

Soil erosion modelled data in mm are presented in Table 8.5., and are converted into g/m$^2$ in Table 8.7., through the field bulk density data in g/m$^3$ in Table 8.6.

Table 8.5. Modelled soil erosion average data per land use and month (mm/month)

| | | Soil erosion per land use (mm) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Year** | **Month** | **DCT** | **SCT** | **DS** | **SS** | **COT** | **AOT** | **CV** | **PTA** | **BS** |
| **1999** | **August** | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| | **September** | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| | **October** | 0.00 | 0.00 | 0.00 | 0.40 | 10.49 | 0.10 | 7.75 | 0.00 | 1.01 |
| | **November** | 0.00 | 0.00 | 0.00 | 0.62 | 16.19 | 0.15 | 7.68 | 0.00 | 1.56 |
| | **December** | 0.00 | 0.00 | 0.00 | 0.01 | 0.23 | 0.00 | 0.17 | 0.00 | 0.02 |
| **2000** | **January** | 0.00 | 0.00 | 0.00 | 0.48 | 8.14 | 0.12 | 6.01 | 0.00 | 1.22 |
| | **February** | 0.00 | 0.00 | 0.00 | 0.03 | 0.73 | 0.01 | 0.54 | 0.00 | 0.07 |
| | **March** | 0.00 | 0.00 | 0.00 | 0.31 | 7.04 | 0.10 | 0.00 | 0.00 | 1.06 |
| | **April** | 0.00 | 0.00 | 0.00 | 0.00 | 1.72 | 0.00 | 1.27 | 0.00 | 0.26 |
| | **May** | 0.00 | 0.00 | 0.00 | 0.00 | 11.94 | 0.00 | 0.00 | 0.00 | 0.00 |
| | **June** | 0.00 | 0.00 | 0.00 | 0.15 | 1.86 | 0.00 | 2.86 | 0.00 | 0.37 |
| | **July** | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |

DCT: dense cork trees; SCT: sparse cork trees; DS: dense shrub; SS: sparse shrub; COT: cultivated olive trees; AOT: abandoned olive trees; CV: cultivated vineyard; PTA: pine trees afforestation; BS: bare soil.

Table 8.6. Soil bulk density measured per land use and month (g/cm$^3$)

| | | Bulk Density (g/cm3) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Year** | **Month** | **DCT** | **SCT** | **DS** | **SS** | **COT** | **AOT** | **CV** | **PTA** | **BS** |
| **1999** | **August** | 0.99 | 1.39 | 1.27 | 1.36 | 1.29 | 1.27 | 1.23 | 1.12 | 1.34 |
| | **September** | 1.14 | 1.20 | 1.26 | 1.38 | 1.36 | 1.21 | 1.42 | 1.14 | 1.41 |
| | **October** | 1.14 | 1.31 | 1.50 | 1.46 | 1.47 | 1.15 | 1.89 | 1.15 | 1.32 |
| | **November** | 1.10 | 1.43 | 1.23 | 1.73 | 1.50 | 1.23 | 1.46 | 1.30 | 1.44 |
| | **December** | 0.86 | 1.18 | 1.13 | 1.23 | 1.17 | 1.04 | 1.52 | 1.03 | 1.31 |
| **2000** | **January** | 1.13 | 1.15 | 1.00 | 1.15 | 1.10 | 1.26 | 1.42 | 1.05 | 1.19 |
| | **February** | 1.05 | 1.22 | 1.33 | 1.18 | 1.37 | 1.08 | 1.86 | 0.80 | 1.16 |
| | **March** | 1.03 | 1.27 | 1.24 | 1.30 | 1.23 | 1.15 | 1.59 | 0.82 | 1.24 |
| | **April** | 0.97 | 1.26 | 1.11 | 1.37 | 1.34 | 1.23 | 1.41 | 0.99 | 1.27 |
| | **May** | 1.06 | 1.16 | 1.20 | 1.28 | 1.41 | 1.18 | 1.49 | 1.18 | 1.33 |
| | **June** | 0.91 | 1.10 | 1.34 | 1.04 | 1.28 | 1.13 | 2.24 | 0.72 | 1.28 |
| | **July** | - | - | - | - | - | - | - | - | - |

DCT: dense cork trees; SCT: sparse cork trees; DS: dense shrub; SS: sparse shrub; COT: cultivated olive trees; AOT: abandoned olive trees; CV: cultivated vineyard; PTA: pine trees afforestation; BS: bare soil.

Table 8.7. Simulated average soil erosion data per land use and month (g/m$^2$)

| | | Soil erosion per land use (g/m2) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Year** | **Month** | **DCT** | **SCT** | **DS** | **SS** | **COT** | **AOT** | **CV** | **PTA** | **BS** |
| **1999** | **August** | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| | **September** | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| | **October** | 0.00 | 0.00 | 0.00 | 0.28 | 7.14 | 0.09 | 4.11 | 0.00 | 0.77 |
| | **November** | 0.00 | 0.00 | 0.00 | 0.36 | 10.79 | 0.13 | 5.24 | 0.00 | 1.08 |
| | **December** | 0.00 | 0.00 | 0.00 | 0.01 | 0.19 | 0.00 | 0.11 | 0.00 | 0.02 |
| **2000** | **January** | 0.00 | 0.00 | 0.00 | 0.42 | 11.53 | 0.10 | 3.19 | 0.00 | 1.03 |
| | **February** | 0.00 | 0.00 | 0.00 | 0.02 | 0.53 | 0.01 | 0.29 | 0.00 | 0.06 |
| | **March** | 0.00 | 0.00 | 0.00 | 0.24 | 5.73 | 0.09 | 0.00 | 0.00 | 0.85 |
| | **April** | 0.00 | 0.00 | 0.00 | 0.00 | 1.28 | 0.00 | 1.40 | 0.00 | 0.20 |
| | **May** | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| | **June** | 0.00 | 0.00 | 0.00 | 0.14 | 1.93 | 0.00 | 1.28 | 0.00 | 0.29 |
| | **July** | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| ***Total average values*** | | *0.00* | *0.00* | *0.00* | *0.13* | *3.56* | *0.04* | *1.42* | *0.00* | *0.39* |

DCT: dense cork trees; SCT: sparse cork trees; DS: dense shrub; SS: sparse shrub; COT: cultivated olive trees; AOT: abandoned olive trees; CV: cultivated vineyard; PTA: pine trees afforestation; BS: bare soil.

### 8.1.6.2. Overland flow simulated data

Simulated overland flow data are presented per land use and month in Table 8.8.

Table 8.8. Predicted overland flow average data per land use and month (mm/month)

| | | Overland flow (mm/month) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Year** | **Month** | **DCT** | **SCT** | **DS** | **SS** | **COT** | **AOT** | **CV** | **PTA** | **BS** |
| **1999** | **August** | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| | **September** | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| | **October** | 0.00 | 0.00 | 0.00 | 12.11 | 15.82 | 10.20 | 15.82 | 0.00 | 12.11 |
| | **November** | 0.00 | 0.00 | 0.00 | 15.72 | 20.54 | 13.24 | 15.72 | 0.00 | 15.72 |
| | **December** | 0.00 | 0.00 | 0.00 | 1.20 | 1.57 | 1.01 | 1.57 | 0.00 | 1.20 |
| **2000** | **January** | 0.00 | 0.00 | 0.00 | 13.57 | 17.73 | 11.43 | 11.43 | 0.00 | 13.57 |
| | **February** | 0.00 | 0.00 | 0.00 | 2.44 | 3.18 | 2.05 | 3.18 | 0.00 | 2.44 |
| | **March** | 0.00 | 0.00 | 0.00 | 10.47 | 12.43 | 10.47 | 0.00 | 0.00 | 12.43 |
| | **April** | 0.00 | 0.00 | 0.00 | 0.00 | 5.32 | 0.00 | 6.95 | 0.00 | 5.32 |
| | **May** | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| | **June** | 0.00 | 0.00 | 0.00 | 6.63 | 6.63 | 0.00 | 8.67 | 0.00 | 6.63 |
| | **July** | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| ***Total average values*** | | ***0.00*** | ***0.00*** | ***0.00*** | ***5.65*** | ***7.57*** | ***4.40*** | ***5.76*** | ***0.00*** | ***6.31*** |

DCT: dense cork trees; SCT: sparse cork trees; DS: dense shrub; SS: sparse shrub; COT: cultivated olive trees; AOT: abandoned olive trees; CV: cultivated vineyard; PTA: pine trees afforestation; BS: bare soil.

### 8.1.7. BASIC APPROACHES TO MODEL VALIDATION

There are many statistical tools for model validation, although here we are going to take into account three common and simple approaches to evaluating how well the model estimates match the observed data. *Absolute difference (AD)*, calculated as the actual difference, where the sign (+ or -) may be an important indicator of performance:

AD = Estimated – Observed (8-1)

*Relative difference*, based on the normalization of the values in order to remove scale effects. Can be expressed as a percentage difference *(RDa)* or as a ratio *(RDb)*:

RDa = ((Estimated – Observed) / Observed) *100 (8-2)

RDb = Estimated / Observed (8-3)

*Correlation:* In regression analysis, an equation is estimated which relates a dependent (or unknown) variable to one or more independent variables. Correlation analysis determines the degree to which the variables are related and for instance how well the estimating equation actually describes the relationship. In the case of model validation, the degree to which observed and estimated values are related is determined. The most commonly used measure of correlation is the coefficient of determination $R^2$, which describes the amount of variation in the dependent variable, which is explained by the regression equation. $R^2$ can range from 0 to 1, with a value of 0 for no correlation and 1 for perfect correlation. Acceptable values of $R^2$ can vary depending on the type of comparison being made, but it would ideally explain more than half of the variation ($R^2$>0.5). Although, a good correlation between modelled and observed data means that the model results vary in a similar way to the observations, it does not necessarily mean that the model results agree with the observations. An $R^2$ of 1 can result even where the modelled results are double the quantity of the equivalent field data. Thus, the correlation coefficient does not measure whether there is systematic under or over prediction.

### 8.1.8. SIMULATED AND OBSERVED DATA ASSESSMENT

#### 8.1.8.1. Analysis per month and land use

Several studies agree that land use and precipitation can greatly affect runoff and soil erosion (e.g. Kosmas et al., 1997; Dunjó et al., 2003; Lasanta et al., 2000). Here, the assessment of the DIS model validation, will be performed on a monthly and land use basis in order to understand both effects.

For all land uses, the observed soil-erosion and overland-flow data are estimated to be highest in September (118 mm), the rainiest month throughout the study period (Table 8.2; Figure 8.1 A). Similarly, soil erosion and overland flow outputs predicted by the DIS model, would be expected also to have high values in September. On the contrary, the model simulates neither soil erosion nor overland flow in this particular month (Figure 8.1 B). To some extent, this could be due to the very low antecedent soil-moisture conditions in summer and in particular in August, the preceding month (Boix-Fayos et al., 1998) (Table 8.2). The soil moisture content is likely to have an important role in the overall model, since it is directly dependent on most of the DIS components.

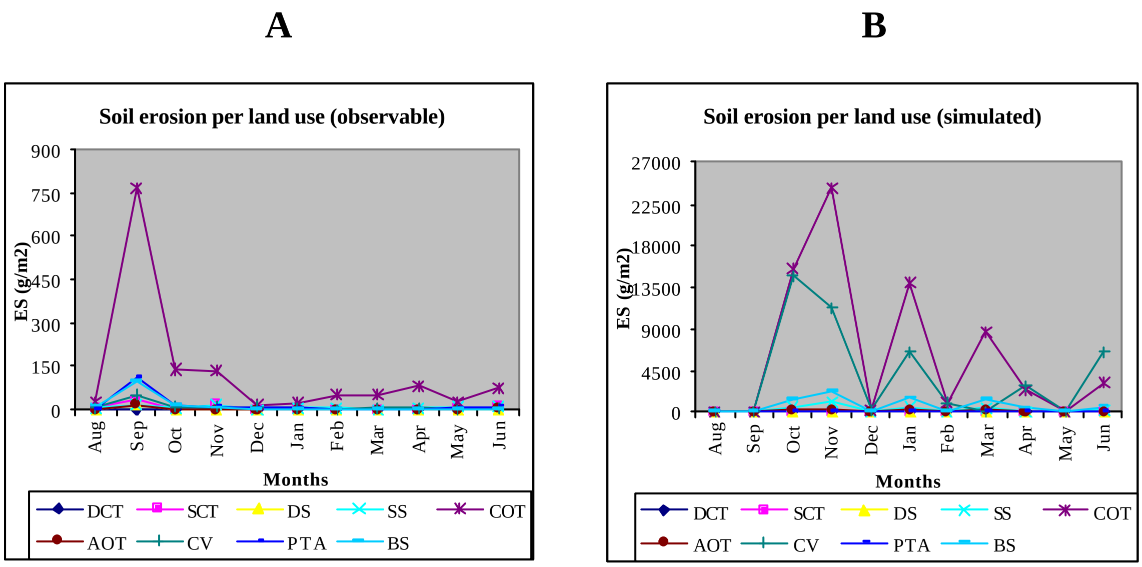


Figure 8.1. Field (A) and predicted (B) soil erosion average data per land use and month

In the DIS template, the runoff water can be either generated by infiltration excess (Hortonian overland flow) or by saturation of the soil and the production of saturation overland flow (Mulligan, 1998). Since the infiltration rate for most soils decreases as the soils are wetted, as the soil saturates the probability of overland flow by infiltration excess increases (Mulligan, 1998). Therefore, when the soil-moisture content is low, potentially the soil infiltration capacity is higher and the runoff generation is likely to be lower than in the case of wetter soils for the same amount of rainfall. The same discussion may be applied to soil erosion by water, since overland flow is reported as a major cause of sediment yield (Mulligan, 1998).

At the catchment scale sensitivity analysis showed that vegetation cover was not the most important factor in controlling runoff and erosion, rather the location of the land use relative to catchment wide flow paths is more important. At the plot scale these processes of connectivity do not operate and vegetation cover seems to take dominance.

The high observed runoff and soil erosion rates determined in poorly vegetated environments such as the cultivated vineyard (CV), cultivated olive trees (COT) and bare soil (BS), are presumably due to the reduction of infiltration capacity, as a result of soil sealing and crusting, processes caused by the raindrop impact on the unstable aggregates in a bare soil (disaggregation) (Porta et al., 1994). According to Mermut et al. (1997), the development of surface seals in cultivated soils (e.g. CV and COT) during rainstorms reduces infiltration rates, increases surface runoff and the erosion hazard, and consequently the loss of organic matter and soil fertility. On the other hand, the development of vegetation such as in those highly vegetated environments with annual vegetation and plant residues such as the dense cork trees (DCT), abandoned olive trees (AOT), and dense shrub (DS), not only prevents the formation of surface sealing but also increases the porosity of the upper soil horizon. Consequently its infiltration capacity is increased and the velocity of runoff water minimized, reducing sediment yield, as a result of direct raindrop impact protection (Kosmas et al., 1997; Romero-Díaz et al., 1999; Boix-Fayos et al., 1998). Vegetation cover has long been recognized as an important factor in runoff generation and soil protection, as vegetation increases infiltration and surface roughness and reduces the kinetic impact of raindrops (Wischmeier et al., 1971; Morgan, 1991; López-Bermúdez et al., 1998; Martínez-Mena et al., 1998). Although in the DIS model, soil crusting and sealing are not directly considered as processes themselves, their effects are indirectly accounted for in the soil-erosion equation process, in which the vegetation cover plays an essential role in the overall output values.

Generally, the model apparently performs better under low or "normal" monthly rainfall amount characteristics such as in January (50 mm), than for periods of high rainfall which, in this environment, are usually characterised by high-energy storm events such as the exceptional one found in September (118 mm). Henceforth, data from September are not evaluated in the model validation, since the data predicted by the DIS model is considered unreliable because at a monthly timestep, the model is not capable of dealing with such high intensity events even though a monte carlo procedure for the allocation of rainfall intensities is use.

**A** **B**

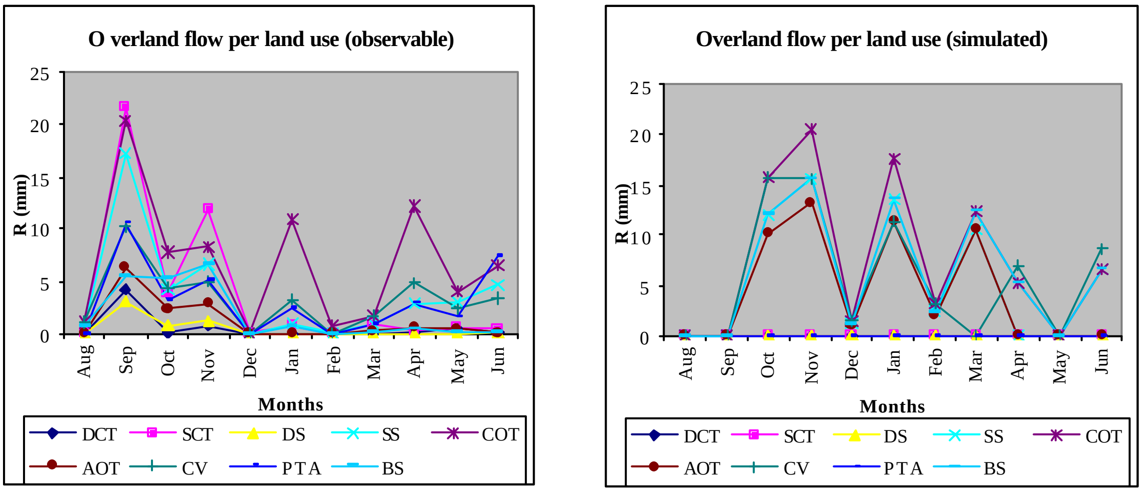


Figure 8.2. Observed (A) and simulated (B) overland flow average data per land use and month

Is obvious from Figures 8.1 A-B that the COT and the DCT environments are the most and least potentially erodible land uses in both cases, respectively. The predicted results show that the second and third most erodible environments are the CV and BS land uses respectively, as could have been anticipated accounting for their low vegetation cover (Table 8.1). Similarly, field data illustrate high soil-erosion rates for both environments (CV and BS), although unexpectedly, the pine trees afforestation (PTA), which has 78% of vegetation cover, is revealed as one of the most erodible environments.

To some extent, the differences between two woodland communities such as the PTA and the DCT in terms of runoff generation and sediment yield, can be related to the response of particular species to interception and stemflow, and to litter production and distribution (Sala and Calvo, 1990). However, only the percentage of vegetation cover has been taken into account in fieldwork campaigns and in the DIS model development. Hence, neither the type of vegetation cover, nor the structure or leaves have been considered. Assuming that the pine-tree leaves have a lower specific area than the DCT ones, there presumably exists a reduction of the rainfall interception, potentially enhancing the overland flow and soil erosion hazard. Furthermore, its hardly mineralizable, decaying debris leads to a slower incorporation of the humic compounds into the soil profile, involving a slow improvement of the soil structure (Dunjó et al., 2003). Likewise, it is well known that soil structure is dependent on soil physico-chemical properties such as soil texture and bulk density but mostly on the soil organic matter content (Tisdall and Oades, 1982). Some of the submodels of the DIS require data of several soil physical properties (e.g. soil texture, soil depth, soil porosity, bulk density) for each land use under study. However, neither soil chemical nor biological properties are considered in the model, since they are out of the scope of this assessment mainly due to their difficult measurement and modelling at a catchment scale. Hence, some of the differences between environments and months of the predicted and observed results may be attributed to the lack of assessment of some of these important parameters (e.g. soil organic matter), which indirectly or directly affect soil-degradation processes.

Figures 8.1 A and B, demonstrate the great difference in simulated and observed soil-erosion rates. In absolute terms, the range of observed and simulated monthly soil erosion values fluctuates from 0 to 766 and from 0 to 24274 g/m$^2$, respectively. The DIS model over-predicts soil erosion for sparse covers such as the sparse shrub (SS), cultivated olive trees (COT), abandoned olive trees (AOT), cultivated vineyard (CV) and bare soil (BS) land uses an unpredicts for full covers such as the dense cork trees (DCT), sparse cork trees (SCT), dense shrub (DS) and pine trees afforestation (PTA), where the DIS model simulates neither soil erosion nor runoff.

According to Nicolau et al. (1996), in semi-arid mountain areas such as the Serra de Rodes catchment, slopes are often rocky and poorly covered with shallow soils, such as the PTA environment located in the uppermost part of the study area (235 m a.s.l.), whereas at the base of the slopes extensive sediment fills develop that are cut by ephemeral-drainage systems. Thus, it seems likely that the stone cover is revealed as an important factor in the generation of the high PTA runoff and erosion rates determined in the field (Table 8.9). Several studies agree that rock-fragment cover at the soil surface have an ambivalent effect on several hydrological processes such as infiltration rate and overland flow, as well as soil erosion rates (Poesen and Lavee, 1994). On the one hand, rocks prevent direct infiltration of raindrop water into the soil and in some cases even produce rock flow resulting in an increase of overland flow volume (Poesen and Lavee, 1994). On the other hand, however, rock fragments cause an increase of water intake rates by protecting the soil surface against raindrop impact forces, and therefore preventing the soil surface from sealing and crusting (Poesen and Lavee, 1994). According to several researchers (Poesen et al., 1990; Valentin, 1994; Yair and Lavee, 1976), the increase or decrease of the total volume of infiltration and overland flow depends on various factors such as position, size and cover of rock fragments a well as structure of the fine earth (Poesen and Lavee, 1994). In relation to soil-erosion processes, direct and indirect effects of rock fragments due to rainfall and runoff reveals that these effects depend on the spatial scale of study as well as to the soil porosity, slope, vertical position and size of rock fragments (Abrahams and Parsons, 1994). Thus, the relation between rock-fragment cover and sediment yield is variable depending on which of these factors are most prominent under given circumstances (Poesen et al., 1994). From the observed results and on the basis of the aforementioned discussion, it seems that although neither fieldwork research nor laboratory experiments have been conducted on the direct and indirect effects of rocky soils, the high stone in the PTA environment (47%), presumably has a positive effect on the overland flow and sediment yield (Table 8.9). As a result, it seems that differences in the cover of rock fragments and vegetation highly affect runoff and sediment yield (Romero-Díaz et al., 1999).

However, predicted results do not reflect the effect of the stoniness cover in any land use test, since this factor has not been considered in the development of the DIS model.

The monthly assessment of soil-erosion data for land use (Figure 8.1 A-B) shows that in general, simulated results are more significantly and positively correlated with monthly total rainfall than the field ones, which are most related in September. Both datasets indicate high values of erosion for all environments in October and November, and although field data have they greatest values concentrated in September (the rainiest month), simulated results have their's in November, the third rainiest month over the study period. Nevertheless, taking into account the months with high rainfall, predicted data are somehow related to the amount of rainfall in October, January, March and June, but not in May, which is the second rainiest month (Table 8.2). Unexpectedly, both datasets (modelled and measured) give for all land uses, very low values of sediment yield in May, where the rainfall amount was very high (75 mm). This could be due to the wet antecedent conditions of the soil during the previous month, April. Therefore, the same aforementioned discussion on the low simulated values in September may apply here as well.

Table 8.9. Stoniness cover percentage per each of the nine land uses

| | DCT | SCT | DS | SS | COT | AOT | CV | PTA | BS |
|---|---|---|---|---|---|---|---|---|---|
| **Stoniness (%)** | 23 | 24 | 17 | 16 | 25 | 20 | 11 | 47 | 15 |

DCT: dense cork trees; SCT: sparse cork trees; DS: dense shrub; SS: sparse shrub; COT: cultivated olive trees; AOT: abandoned olive trees; CV: cultivated vineyard; PTA: pine trees afforestation; BS: bare soil.

Simulated and observed data show very low soil-erosion values in some of the less rainy months (August and December), and also in May (Table 8.3 and 8.7) for all land uses. Predicted soil erosion results are zero in August, September and in May for all land uses (Table 8.7).

With reference to the predicted and field overland flow data, it is clear from the charts that both datasets are in the same value interval, ranging from 0 to 21 mm, although data follow neither the same monthly nor the same land-use pattern (Figure 8.2 A-B). As for the soil-erosion output, the model does not simulate overland flow in any month for the DCT, SCT, DS and PTA land uses, presumably accounting, as mentioned above, for the importance of the vegetation cover in the DIS model (Table 8.8). Likewise and, as for soil erosion, the model predicts no runoff in August, September or in May for all land uses (Figure 8.2 B). This results from the high interdependence of soil erosion and overland-flow data in the model, since the overland flow is one of the main components of the soil erosion sub-model. From the field data, in general the COT is the land use which generates runoff water in the most months (October, January, February, April, and May). However, the SCT has the highest runoff of the overall study period in November, where it is slightly greater than the COT environment. The SS and PTA environments, which are quite vegetated (50 and 78% respectively), have significant overland-flow values in some months (e.g. September). As expected, the CV and BS have high values and the DCT and DS may be considered the ones that generate the least overland flow in each assessed month (Figure 8.2 A).

The SCT is one of the environments where the DIS model generates neither erosion nor overland flow in any of the studied months. However field data determinations reveal the SCT as a land use with very high overland flow and soil loss in some of the assessed months (e.g. September and November). The vegetation cover set for the SCT in the DIS and determined in the field, is the lowest (30%) of the group of land uses where the model generates zero output values (DCT: 79%; DS: 80% and PTA: 78%). Therefore, these identical simulated results do not result from the effect of vegetation-cover protection, but from other factors shared by all of these land uses. Thus, looking at the field data used as input in the DIS model (Table 8.1), the highest saturated hydraulic conductivity (SatHK) values are revealed in the SCT, DCT, DS, and PTA land uses. The SatHK, calculated on the basis of the soil textural class, bulk density and soil porosity, expose these environments as the most permeable ones and so, the ones with less potential to be eroded or to generate runoff water (Martínez-Fernández et al., 1995). Conversely, the most potentially erodible environments such as the COT, CV and BS have the lowest SatHK values, being the most impermeable land uses. The important role of this parameter is also corroborated by the sensitivity analysis, which has determined the SatHK as the most sensitive variable of the overall set of inputs tested (see chapter seven). Thus, accounting for its high sensitivity, the determination of the SatHK should be as accurate and precise as possible. An alternative way of calculating its value would be on the basis of the field determinations of the infiltration capacity of each land use, which should be more reliable than the values calculated in the HYDROPAR model.

### 8.1.8.2. Analysis for land use of the overall study period

For this analysis, soil erosion and overland flow totals have been grouped according to each land use under assessment. Simulated and observed data from both outcomes are displayed in Figure 8.3 (A) and (B) respectively.

A B

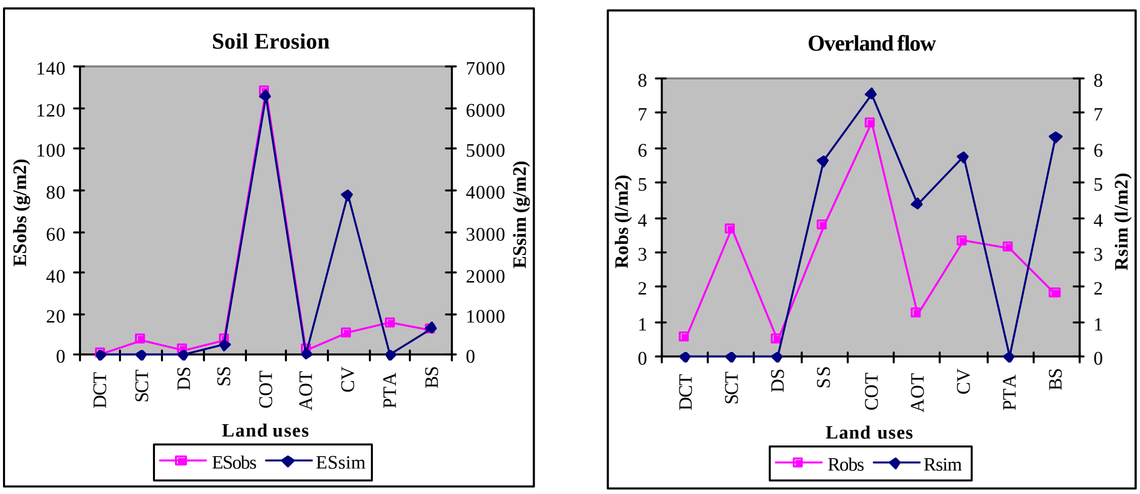


Figure 8.3. Simulated and observed soil erosion (A) and runoff (B) average data per land use of the overall study period

Regarding soil-erosion data, it is clear from the chart (Figure 8.3. A), that simulated data are higher than those determined in the field, with the exception of the dense cork trees (DCT), sparse cork trees (SCT), dense shrub (DS) and pine trees afforestation (PTA) land uses, where the model predicts null data (Table 8.7).

These environments (DCT, SCT, DS and PTA) with vegetation cover percentages of: 79, 30, 80, and 78%, respectively, have the highest values of saturated hydraulic conductivity (SatHK) (Table 8.3), the most important variable assessed in the Sensitivity Analysis of the DIS model (chapter seven). It seems from the results that when some intrinsic land use characteristics such as the saturated hydraulic conductivity (SatHK), have high values, model outputs are less or not dependent on the monthly amount and/ or intensity of rainfall. This is particularly obvious when assessing the month of September (Table 8.4), where the model neither generates runoff nor sediment yield in these land uses, although this month has the greatest rainfall recorded amount (118.57 mm) and it is considered itself very high for the area of study and therefore, for the conditions which where taken into account when developing the DIS model.

**A** **B**

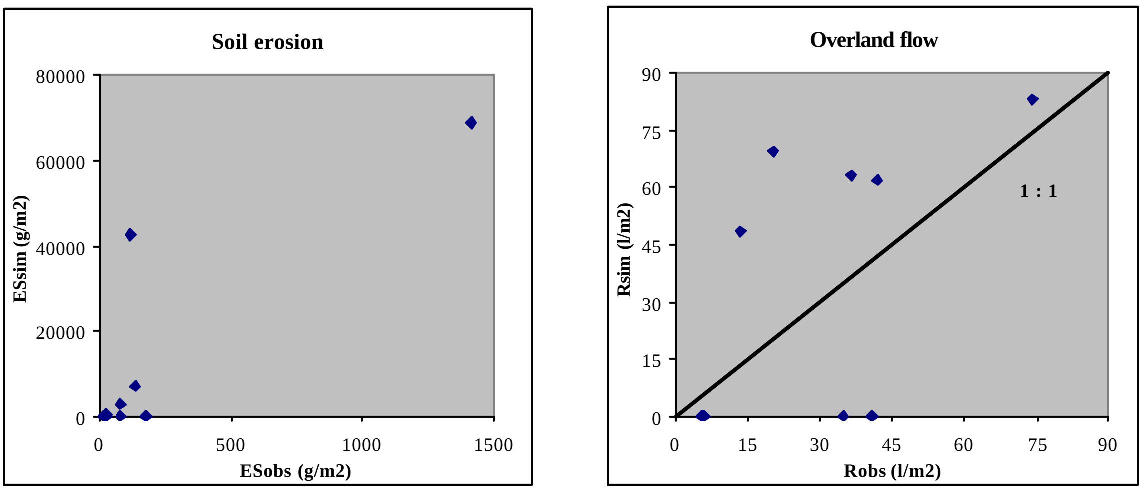


Figure 8.4. Simulated versus field soil erosion (A) and runoff (B) average land use data

For the whole period significant soil erosion rates were determined in the field for each of the nine studied land uses, including the DCT, SCT, DS and particularly the PTA, which unexpectedly, after the cultivated olive trees environment (COT), has the greatest total soil erosion rate (175 g/m$^2$) (Table 8.3). Both simulated and observed total soil erosion data, highlight the COT as the most erodible land use, although the predicted data is higher leading to an absolute difference value of 6148 (Table 8.10).

Absolute and relative differences between simulated and observed soil-erosion and overland-flow data for each of the assessed land uses are presented in Table 8.10. With reference to these tests, similar results between simulated and observed data are found in the DCT, DS, SCT, and PTA, in decreasing order. These are the land uses where the model calculates zero values and therefore the ones where the field values are higher than the predicted ones, giving negative values of AD (Table 8.10). The AOT land use also presents quite small AD (51), although in this case, predicted data is greater that the determined in the field. The rest of the environments have high values of AD ranging from 51 for the AOT to 3875 for the CV. It seems that the maximum divergences between values are on the cultivated (COT and CV) and the bare soil (BS) environments and the minimums are concentrated on those land uses of later abandonment and for instance more vegetated (DCT, DS, SCT and PTA).

Table 8.10. Absolute and relative differences between simulated and observed data

| | Soil Erosion (g/m$^2$) | | | Overland flow (l/m$^2$) | | |
|---|---|---|---|---|---|---|
| **Land use** | **AD** | **RDa** | **RDb** | **AD** | **Rda** | **RDb** |
| **Dense cork trees** | -1.20 | -100 | 0 | -0.56 | -100 | 0 |
| **Sparse cork trees** | -7.25 | -100 | 0 | -3.70 | -100 | 0 |
| **Dense shrub** | -2.32 | -100 | 0 | -0.50 | -100 | 0 |
| **Sparse shrub** | 247.95 | 3375 | 35 | 1.86 | 49.00 | 1.49 |
| **Cultivated olive trees** | 6148.32 | 4785 | 49 | 0.85 | 12.62 | 1.13 |
| **Abandoned olive trees** | 50.77 | 2103 | 22 | 3.17 | 259.00 | 3.59 |
| **Cultivated vineyard** | 3874.78 | 36345 | 364 | 2.43 | 73.23 | 1.73 |
| **Pine trees afforestation** | -16.00 | -100 | 0 | -3.14 | -100 | 0 |
| **Bare soil** | 648.16 | 5187 | 53 | 4.48 | 244.44 | 3.44 |

AD: absolute difference; RDa: percentage of difference; RDb: ratio of difference.

In relation to overland flow almost the same above-mentioned discussion may be applied (Figure 8.3 B). Thus, in those environments where the DIS model does not generate runoff, such as the DCT, SCT, DS and PTA, field data were recorded (Table 8.4). However, both types of data are in the same order of magnitude and therefore differences among them are smaller than the ones related to soil erosion (Table 8.10). The differences are much lower than for soil erosion. Assessing runoff AD of the overall set of land uses, the highest value is associated to the BS environment (4.48), followed by the SCT (-3.70), the AOT (3.17) and the PTA (-3.14), whereas the lowest values are related to the DS (-0.50), DCT (-0.56) and COT (0.85) (Table 8.10). Intermediate AD values of 1.86 and 2.43 are related to the SS and CV, respectively. Obviously, in this case also those environments where the DIS model does not simulate results (DS, DCT, SCT and PTA), are those with a negative AD meaning that field results are higher than the predicted ones. For the rest of land uses, the model always predicts higher values of overland flow than the observed ones (Table 8.10). As a result, it seems that the range of overland flow predicted and field values is smaller than the one of soil erosion. This is because soil erosion is runoff to a power so if there is a difference in runoff then soil erosion will be that difference multiplied by the power.

Furthermore, linear correlations have been performed between the simulated and observed soil erosion (Figure 8.4 A) and overland flow (Figure 8.4 B) data. In relation to overland flow data, non-statistically significant results are presented, with a $R^2$ of 0.27, considered very low (Figure 8.4 B). Assuming that the measurements are a good representation of the effect of land use, for runoff the effect of land use is not predicted well. On the other hand, regarding soil erosion, a correlation coefficient ($R^2$) of 0.72 was obtained when taking into account the overall set of land uses, and of 0.9851 when removing the CV land use data. Both cases show very high and positive statistically significant results. However, the previous results highlight how poor the correlation coefficient ($R^2$) is as an indicator of model performance given that at least of estimates are the right order of magnitude.

Hence, it seems that for soil erosion the pattern of change for different land uses and plots is correct, even though the values modelled are much higher than those measured. This indicates that the role of land use is accounted for well but that there is some generic process that is overpredicting soil erosion. May be the overestimation of the soil loss could be due to the fact that deposition/transport capacity is not taken into account in the model (see section 8.1.8.4.).

The overestimation of soil erosion could also be explained on the basis of the different kind of erosion measured in the microplots and simulated by the DIS. The eroded soil measured in the microplots may be mainly related to splash erosion, although to some extent it also accounts for interrill erosion, whereas soil erosion modelled by the DIS is basically associated to runoff water. Thus, we could expect different soil erosion results from both determinations. Furthermore, it seems that a scaling issue may also account for the overestimation of soil erosion, since the measured data relates to 1m$^2$ (area of a microplot) and the simulated to a 25 m$^2$ (1 pixel area). Thus, the model predicts 25 times more discharge (runoff) in 1 single pixel of the model, than in 1 m$^2$ area of the microplot. Therefore, field results should be scaled to 25 m$^2$ in order to be equivalent to the modelled area. In addition, since overland flow to the power of two, is a component of the soil erosion equation, the modelled soil erosion could be expected to be 625 times ($25^2$) higher than the measured ones. As a result, taking into account the scaling factor, the outcomes of the model seem more reasonable.

#### 8.1.8.3. Linear correlation analyses for all land uses: seasonal and total

Monthly average values of simulated and observed soil erosion and overland flow are presented in Table 8.11. For both outcomes, field results show their maximum and minimum value in September and December, respectively, the rainiest and least rainy month of the study period. On the other hand, predicted data have their maximum in November, the third rainiest month, whereas the minimum is shared by August, September and May, the three months for which the model simulates zero values. For all linear correlation analyses performed, data related to the month of September have been excluded, since its rainfall is considered of torrential nature and the model is inappropriate for these kinds of conditions, which would lead to a mistaken assessment of the rest of dataset.

The first type of linear correlations were performed taking into account data of all land uses for each month. No correlation was found for the months where the model neither simulates erosion nor runoff, which are: August, September and May. In general, highly statistically significant correlations are found for the soil erosion data (Table 8.12).

Table 8.11. Monthly simulated and field average soil erosion and runoff data

| | Soil erosion (g/m2) | | Overland flow (mm) | |
|---|---|---|---|---|
| **Month** | **Observed** | **Simulated** | **Observed** | **Simulated** |
| **August** | 7.32 | 0.00 | 0.47 | 0.00 |
| **September** | 124.17 | 0.00 | 11.00 | 0.00 |
| **October** | 22.28 | 3566.08 | 3.61 | 7.34 |
| **November** | 21.84 | 4335.28 | 5.41 | 8.99 |
| **December** | 4.82 | 62.53 | 0.02 | 0.73 |
| **January** | 5.50 | 2502.30 | 2.13 | 7.53 |
| **February** | 6.13 | 237.37 | 0.11 | 1.48 |
| **March** | 7.09 | 1164.23 | 0.76 | 5.09 |
| **April** | 11.07 | 603.97 | 2.74 | 1.95 |
| **May** | 5.72 | 0.00 | 1.46 | 0.00 |
| **June** | 14.05 | 1133.26 | 2.61 | 3.17 |

However, performing the same analysis excluding the cultivated vineyard environment (CV), new $R^2$ are highly statistically significant, ranging from 0.91 to 0.99. Hence, it seems that predicted soil erosion data are highly associated with the observed ones for all the assessed months and land uses excepting the CV environment, although both dataset ranges are extremely different (see section 8.1.8.2) (Table 8.12).

Concerning overland flow results, monthly linear correlations are not as significant as the erosion ones. Not only are $R^2$ values lower but they are also fewer months were they are significant (Table 8.12). Here, even when excluding some land use data in order to obtain a better correlation in a particular month, results are not always as significant as for soil erosion (Table 8.12). Considering data of all land uses, only the month of October shows significantly positive correlation between the two sets of runoff data ($R^2$ = 0.52). The rest of significant correlations presented in bold in Table 8.12 correspond to tests performed without runoff data of one or more land uses.

Table 8.12. Linear correlations of simulated (y) and measured (x) soil erosion and runoff monthly land use data

| | Soil erosion (g/m²) | | Overland flow (l/m²) | |
|---|---|---|---|---|
| **Months** | **Equation** | **$R^2$** | **Equation** | **$R^2$** |
| **August** | - | - | - | - |
| **September** | - | - | - | - |
| **October** | y = 0.005x + 4.4654 | **0.5114** | y = 0.2311x + 1.9164 | **0.5197** |
| **November** | y = 0.0046x + 2.0973 | **0.8074** | y = 0.0981x + 4.526 | 0.0606 |
| | | | *(d) y = 0.3266x + 0.747* | **0.8486** |
| **December** | y = 0.0594x +1.1038 | **0.8107** | y = 0.0003x + 0.0227 | $2x10^{-5}$ |
| **January** | y = 0.0014x + 2.1108 | **0.8917** | y = 0.254x + 0.226 | 0.2908 |
| | | | *(e) y = 0.3312x – 0.2605* | 0.4334 |
| **February** | y = 0.0251x + 0.1707 | 0.4226 | y = 0.0921x – 0.0303 | 0.23 |
| | *(a) y = 0.0507x – 0.2574* | **0.9897** | *(f) y = 0.1603x – 0.0397* | **0.5186** |
| **March** | y = 0.0061x –0.0073 | **0.9644** | y = 0.008x + 0.7146 | 0.0053 |
| | | | *(g) y = 0.0701x – 0.002* | 0.3771 |
| **April** | y = 0.0145x + 2.3263 | 0.3442 | y = 0.7595x + 1.2536 | 0.3257 |
| | *(b) y = 0.0358x + 0.2323* | **0.9706** | *(h) y = 1.0766x + 1.372* | **0.5626** |
| **May** | - | - | - | - |
| **June** | y = 0.0032x +10.421 | 0.0864 | y = 0.2648x + 1.7702 | 0.1135 |
| | *(c) y = 0.0221x + 4.8338* | **0.9147** | *(i) y = 0.4579x + 0.3701* | 0.4841 |
| | | | *(j) y = 0.5791x + 0.4572* | **0.7619** |

y= modelled data; x= measured data.
Data analysis per land use with the exception of (a) = cultivated vineyard; (b) = cultivated vineyard; (c) = cultivated vineyard; (d) = pine trees afforestation and sparse cork trees; (e) = pine trees afforestation and abandoned olive trees; (f) = cultivated vineyard and bare soil; (g) = cultivated vineyard, sparse cork trees and pine trees afforestation; (h) = bare soil; (i) = pine trees afforestation; (j) = pine trees afforestation and bare soil. In bold: significant correlations ($R^2$>0.5).

Best correlations range from 0.52 to 0.85, as compared to the soil erosion results, only five of the ten studied months present statistically significant correlations. Here, the land use that is more commonly excluded from the correlation analysis is the pine trees afforestation environment (PTA). Other excluded environments are the sparse cork trees (SCT), the abandoned olive trees (AOT) and also the CV and the bare soil (BS) (Table 8.15). In general it is likely that the DIS model predicts less reliable results when computing soil erosion for the CV land use, and also when simulating overland flow for the PTA environment.

The second linear correlation test has been performed by grouping the several months under study in two different sets according to their amount of rainfall. The low rainfall amount group includes: August (Aug), December (Dec), February (Feb), April (Apr) and June (Jun), with rainfall amounts ranging from 5 to 25 mm. On the other hand, the high rainfall group, which comprises: October (Oct), November (Nov), January (Jan), March (Mar) and May (May), has amounts ranging from 45 to 75 mm. Here, soil erosion data are better correlated than those from overland flow, for both groups under study. Soil erosion regressions for the overall set of land uses of the several months of high rainfall, are presented in Table 8.13, with a $R^2$ of 0.61.

Table 8.13. Regression of soil erosion (y) against runoff (x), grouping several months according to their rainfall amount and taking into account data of all land uses

| | **Low rainfall months (Aug-Dec-Feb-Apr-Jun)** | | **High rainfall months (Oct-Nov-Jan-Mar-May)** | |
|---|---|---|---|---|
| **Variables** | **Equation** | **$R^2$** | **Equation** | **$R^2$** |
| **Soil erosion ($g/m^2$)** | y = 0.0278x + 3.7327 | **0.8139** | y = 0.0043x + 2.6413 | **0.6081** |
| **Overland flow (mm)** | y = 0.7295x + 0.2981 | **0.5034** | y = 0.2496x + 0.9249 | **0.4389** |

Likewise, the correlation for the low rainfall months is even higher ($R^2$ = 0.81), although in this case data related to the cultivated vineyard (CV) land use from Feb, Apr and Jun, have been excluded from the analysis. Overland flow linear correlation related to the high rainfall group, has a $R^2$ of 0.44, excluding data of the PTA and SCT environments from November and March, of the AOT from January, and of the CV and PTA from March. Statistically significant correlation has been found for the low rainfall group ($R^2$ = 0.50), when excluding data of the BS, AOT and CV from February, of the BS from April and of the PTA and BS land uses from June (Table 8.13). Thus, it is quite clear from the results that even when removing some land use data from the analysis in order to obtain better correlations, these are always better for soil erosion than for overland flow. In conclusion, and on the basis of the exposed linear relationships between data predicted by the model and data determined in the field, the DIS seems to perform slightly better for both outcomes (erosion and runoff), when the monthly rainfall amount is relatively low (Table 8.13). This means that the model probably does not deal well with high rainfall intensities and the hortonian runoff generation mechanism, probably a result of the timestep used.

The third and final of regression has been conducted for soil erosion and runoff data of the overall set of land uses and considering together all months of the study period, but September (Table 8.14). Table 8.14 shows a $R^2$ of 0.47 and 0.25 for soil erosion and overland flow respectively. Therefore, neither soil erosion nor overland flow correlations are statistically significant. However, since the soil erosion correlation coefficient has almost a value of 0.5, to some extent it could be considered significant, considering the large set of data analysed (N = 90).

Table 8.14. Soil erosion and runoff linear correlations of the overall set of land uses and months throughout the overall study period

| Soil erosion (g/m$^2$) | | Overland flow (mm) | |
|---|---|---|---|
| **Equation** | **$R^2$** | **Equation** | **$R^2$** |
| y = 0.0042x 4.8515 | 0.47 | y = 0.244x + 1.0458 | 0.25 |

### 8.1.8.4. The soil erosion transport capacity factor

The DIS model predicts spatial distribution of erosion rates on the basis of each selected land use in the study catchment. The model, accounts for neither sediment transport nor sediment deposition but only for the overall soil loss through the wash erosion model described by Thornes (1990). This model seems to overestimate soil erosion when being validated at a point scale with microplot field data. On the other hand, the field data inherently consider sediment transport and deposition since the measurement is of net sediment flux out of the microplot. This produces much lower values than the wash model. Thus, to some extent, this would explain higher soil erosion rates from the model than measured in the field. Moreover, there are other factors affecting erosion in the field that are not incorporated in the model, including the reduction in erosion likely to accrue from soil surface armouring and sealing, not taken account of in the model.

$$STC = (OFLOW)^{1.7} * \sin(Slope) * 0.01 \tag{8-4}$$

STC: sediment transport capacity (mm/month)
OFLOW: overland flow (mm/month)
Slope: slope angle (degrees)

In order to test whether the calculation of the sediment transport capacity and indirectly of sediment deposition would make any difference in the DIS soil erosion results at a plot scale, the equation for transport capacity described by Kirkby (1976) (Equation 8.4), has been included in the model and new validation results are presented in Table 8.15.

Table 8.15. Change on the final soil erosion modelled data due to the sediment transport capacity calculation: comparison with the observed data

| **Original simulated soil erosion, sediment transport capacity, final soil loss and measured soil erosion ($g/m^2$)** | | | | | **Differences between modelled and observed soil erosion** | | |
|---|---|---|---|---|---|---|---|
| **Land uses** | **OESsim** | **STC** | **FESsim** | **ESobs** | **AD** | **RDa** | **RDb** |
| **DCT** | 0 | 0 | 0 | 1 | -1 | -100 | 0 |
| **SCT** | 0 | 0 | 0 | 7 | -7 | -100 | 0 |
| **DS** | 0 | 0 | 0 | 2 | -2 | -100 | 0 |
| **SS** | 239 | 128 | 128 | 7 | 121 | 1644 | 17 |
| **COT** | 6194 | 272 | 272 | 129 | 144 | 112 | 2 |
| **AOT** | 52 | 93 | 52 | 2 | 50 | 2062 | 22 |
| **CV** | 3694 | 159 | 159 | 11 | 149 | 1395 | 15 |
| **PTA** | 0 | 0 | 0 | 16 | -16 | -100 | 0 |
| **BS** | 658 | 154 | 154 | 13 | 142 | 1136 | 12 |
| **Total average values** | 1204 | 90 | 85 | 21 | 64 | 307 | 4 |

ESsim: original simulated soil erosion; STC: sediment transport capacity; FESsim: final simulated soil erosion; ESobs: observed soil erosion; AD: absolute difference; RDa: percentage of difference; RDb: ratio of difference

Soil erosion is the sediment transport capacity and sediment entrainment (Table 8.15). As a result, the new soil loss predicted data though it is still higher for the same previously discussed land uses, it is much closer to the field results (Table 8.15). Using this approach, the new soil loss predictions though still higher than the field data are much closer to the field results than the model results without considering sediment transport (Table 8.15). Data of the absolute (AD) as well as relative differences (RDa, RDb) between both datasets are presented in Table 8.15 and are always lower than the equivalent without transport limitation. Thus, the "sediment transport capacity" has arisen as a possible additional factor to take into account in the DIS template in order to model better the soil erosion data at a point scale.

#### 8.1.8.5. General model validation conclusions

The first conclusion that may be drawn from the validation analysis is that the DIS model greatly overestimates soil erosion in the cultivated fields (COT and CV). These environments, which are the least vegetated in the study area, are considered to be recently abandoned. Likewise, the model also overestimates soil loss in the bare soil (BS) and abandoned olive trees (AOT) environments, although the differences are lower than for the cultivated ones when compared to the field data. Both predicted and observed data indicate that COT is the most potentially erodible land use. The model underestimates soil erosion in the most vegetated environments such as the dense cork trees (DCT), sparse cork trees (SCT), dense shrub (DS), and pine tree afforestation (PTA), land uses of early abandonment in the catchment. However, the absolute difference between simulated and field data of this group of environments is minimal in comparison to the cultivated ones. In general, although data is not in the same range of values, the linear regressions performed reveal highly significant correlations between simulated and observed soil erosion data. Irrespective of this, the most and least potentially erodible land uses were determined and simulated in the same order, with the exception of the PTA environment, which was assessed as the second most erodible environment in the field. Therefore, predicted data related to the PTA land use, may be considered not very reliable.

In relation to overland flow, both datasets (modelled and measured) have a similar range of values, although linear regressions between them have shown less significant results than for soil erosion. The model overestimates and underestimates overland flow in the same land uses as for soil erosion, although in this case the absolute difference in all land uses is so small that it could be considered negligible. Another significant difference has been found in the SCT land use, which simulated very low overland flow in contradiction to the high values determined in the field. Therefore, the prediction of SCT data by the model, is unlikely to be close to the real values.

In general, results produced by the DIS model seem not very reliable from a quantitative point of view and some adjustment may be needed and applied in future research work. However, results seem to be significant in terms of which land uses are the most and least potentially degraded, and therefore which scenarios would be better for preventing land degradation. In this way the model fulfils its objective as a desertification support tool as it identifies the patterns of change expected, if not the magnitudes. The model would need to be more complex, have better and more input data and a regional scale validation if the magnitudes were to be predicted reliably.

## 8.2. LAND USE SCENARIO OUTCOMES

The DIS model has been created for identifying present and future threats to allow the development of effective measures to combat desertification. Thus, the ultimate part of the creation of the DIS model will be accomplished by the development of several scenarios representing the application of particular policies addressed to land degradation issues.

### 8.2.1. SELECTION OF THE LAND-USE SCENARIOS

The generalised abandonment of cultivated areas in the Mediterranean region during the last century (Gallart and Llorens, 2002; Dunjó et al., 2003), together with the Agricultural Policy of the European Union, has encouraged the withdrawal of agricultural fields, which has, in turn, completely transformed the Mediterranean landscape (Lasanta et al., 2000). As a result of this farmland set-aside, in semi-arid abandoned terrains such as the Serra de Rodes catchment, land degradation has become a common phenomenon, where physical soil loss by water erosion is identified as a dominant problem (Thornes, 1995; Pardini et al., 2003). Furthermore, although forest and bush fires are known to have occurred in the Mediterranean belt throughout historic and prehistoric times (Naveh, 1990), their frequency has considerably increased as a result of this abandonment, resulting in the degradation of larger areas (Figure 1.3) (Dunjó et al., 2003; Cerdà et al., 1995). Accordingly, both land-use change and wildfires, are believed to constitute a major environmental problem in the Serra de Rodes catchment.

Thus, the policies of particular interest in the region are (a) land abandonment and (b) methods for fire prevention and control through the management of vegetation characteristics. Both of these have poorly known implications for the hydrology and ecology of this complex and heterogeneous environment. These scenarios should enable the prediction of potential desertification risk and the assessment of alternatives for management (Stocking and Murnaghan, 2001).

#### 8.2.1.1. General traits

Two different land-use scenarios have been designed. The first one, S1, intends to symbolise the landscape of the overall Serra de Rodes catchment after a hypothetical wildfire.

Table 8.16. Original landscape, wildfire and cultivated fields scenarios characteristics

| Land use, vegetation cover and percentage of total area distribution | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| *Original Scenario (S0)* | | | *Wildfire Scenario (S1)* | | | *Cultivated fields Scenario (S2)* | | |
| **Land use** | **% area** | **%VC** | **Land use** | **% area** | **%VC** | **Land use** | **% area** | **%VC** |
| DCT | 15 | 79 | DCT | 15 | 10 | DCT | 15 | 80 |
| SCT | 11 | 30 | SCT | 11 | 10 | COT | 11 | 25 |
| DS | 25 | 80 | DS | 25 | 10 | CV | 25 | 20 |
| SS | 15 | 50 | SS | 15 | 10 | COT | 15 | 25 |
| COT | 4 | 20 | COT | 4 | 25 | COT | 4 | 25 |
| AOT | 14 | 69 | AOT | 14 | 10 | COT | 14 | 25 |
| CV | 4 | 14 | CV | 4 | 20 | CV | 4 | 20 |
| PTA | 8 | 78 | PTA | 8 | 10 | CV | 8 | 20 |
| ROAD | 3 | 0 | ROAD | 3 | 0 | ROAD | 3 | 0 |
| BS | 1 | 40 | BS | 1 | 10 | CV | 1 | 20 |

DCT: dense cork trees; SCT: sparse cork trees; DS: dense shrub; SS: sparse shrub; COT: cultivated olive trees; AOT: abandoned olive trees; CV: cultivated vineyard; PTA: pine trees afforestation; BS: bare soil; VC: vegetation cover percentage

The second one, S2, is focused on a landscape mostly covered by cultivated fields, which are commonly considered as fire breaks in terms of protection to fire spread. The set of monthly rainfall data selected for the running of the model under these two particular scenarios corresponds to a dry average rainfall year, which has a total annual amount of 430 mm (Table 5.4). Characteristics of the land-use composition, percentage of the total area covered by each land use and the correspondent vegetation cover percentage of each scenario are presented in Table 8.16.

### 8.2.1.2. Land-use scenario after a wildfire (S1)

The most direct appreciable change in the landscape after a wildfire is the drastic reduction of the vegetation cover percentage. In the wildfire scenario (S1), the area covered by the DCT, SCT, DS, SS, PTA, BS and AOT land uses, comprises 88% of the whole study catchment and has been set to 10% of vegetation cover in the model script. The COT, CV and ROAD environments cover the 4%, 4% and 3% of the total landscape, and have a 25%, 20% and 0% of vegetation cover, respectively (Table 8.16). Regarding the rest of the soil characteristics required as input variables in the DIS model, they have been considered the same as for the original landscape (Table 8.1 and

8.16). On the one hand, this is justified by the non-availability of data after a wildfire for the study catchment. On the other hand, by the fact that although most of the soil physicochemical variables and soil behaviour are modified instantaneously after a wildfire and then reverted (Torri and Borselli, 2000), here, it is assumed that these changes are progressively built up over a larger period and not immediately after the disturbance, as the focus of this assessment.

#### 8.2.1.3. Land-use scenario mostly related to cultivated fields (S2)

Concerning the generalised abandonment of the terrains of the study area, now mainly dominated by shrubs and so under a great risk of wildfire, the cultivated fields scenario (S2) aims at finding an agricultural management alternative to the present landscape, with traditional crops of the area such as vineyards and olive trees, by assessing whether it would prevent land degradation or not.

For this scenario, apart from the area originally covered by the road (3%) and the DCT (15%) land uses, the rest have been transformed either into olive trees (COT) or vineyards (CV), composing the 44 and 37% of the overall catchment and with a 25 and 20% of vegetation cover respectively (Table 8.16).

### 8.2.2. COMPARISON OF LAND-USE SCENARIOS

Comparing the two land use scenario outcomes (S1, S2) with the original landscape of the Serra de Rodes catchment (S0), the cultivated scenario is the most potentially erodible and more prone to overland flow (Table 8.17). In fact soil erosion in S2 is almost double than for S0 and S1, and rates are even greater when assessing overland flow (Table 8.17), presumably accounting for the high presence of olive trees (COT) which, as assessed in the validation is the most degradable land use in the study site.

Table 8.17. Original (S0), wildfire (S1) and cultivated (S2) scenarios: comparison of average monthly and total soil erosion and overland flow data (in mm)

| | Original Scenario (S0) | | Wildfire Scenario (S1) | | Cultivated Scenario (S2) | |
|---|---|---|---|---|---|---|
| **Months** | **Soil Erosion** | **Overland flow*** | **Soil erosion** | **Overland flow*** | **Soil erosion** | **Overland flow*** |
| **January** | 0.93 | 172 | 1.17 | 115 | 1.59 | 364 |
| **February** | 1.21 | 387 | 2.18 | 370 | 2.22 | 865 |
| **March** | 1.17 | 162 | 1.05 | 93 | 1.67 | 267 |
| **April** | 1.73 | 487 | 2.24 | 487 | 3.49 | 1216 |
| **May** | 0.52 | 87 | 0.27 | 85 | 0.94 | 211 |
| **June** | 2.75 | 604 | 2.94 | 563 | 4.02 | 1359 |
| **July** | 0.04 | 19 | 0.01 | 24 | 0.02 | 59 |
| **August** | 0.03 | 59 | 0.06 | 112 | 0.07 | 264 |
| **September** | 0.003 | 9 | 0.001 | 15 | 0.003 | 35 |
| **October** | 0.06 | 110 | 0.15 | 223 | 0.18 | 530 |
| **November** | 0.23 | 41 | 0.17 | 53 | 0.25 | 141 |
| **December** | 0.41 | 210 | 0.44 | 198 | 0.50 | 479 |
| **Total average** | **0.76** | **196** | **0.89** | **195** | **1.25** | **482** |

** Units in thousands*

Thus, although cultivated fields are believed to prevent the spread of wildfire, it seems that in the case of poor or no agricultural management, as in the case of the assessed fields, they are at risk of intensive land degradation in terms of soil loss and runoff generation.

Simulated soil erosion data are lower in the original scenario than in the wildfire one, whereas overland flow values are lower in the latter (Table 8.17). Thus, when running the DIS model at a catchment scale, apparently the vegetation cover percentage, which is the only different variable between S0 and S1, seems not as important as it is at a point scale (as was shown in the validation). This actually confirms that although vegetation is believed to influence physical erosion both directly, by increasing surface resistance to wash erosion, and indirectly, by influencing, runoff, and evapotranspiration (Tucker and Bras, 1999), in the DIS model it is not one of the most significant factors, as revealed in the sensitivity analysis of the model (chapter seven).

Both land use scenario outcomes show greater soil-erosion and runoff rates than the ones modelled for the original landscape (S0). On the one hand, results from the wildfire scenario (S1), suggest that wildfire has little effect on land degradation processes when assessing the overall catchment.

According to the FAO/UNEP/UNESCO (1979), soil loss is considered the main indicator of soil erosion by water. Therefore, in order to determine the level of soil degradation under each of the developed land use scenarios (S0, S1 and S2), their respective annual soil erosion has been evaluated on the basis of the Table 3.17 (chapter three). According to this land degradation classification, the three scenarios are considered under moderate land degradation (0.6-3.3), being the current landscape and the cultivated scenario, the least and most potentially degradable scenarios. Thus, it seems that a completely cultivated landscape would not be an effective alternative of sustainable management in the Serra de Rodes catchment, although it would considerably reduce the risk of fire. On the basis of the previous land degradation classification, the simulated soil erosion results at a catchment scale seem to be the right order of magnitude of soil erosion values.

So, even though these two examples of land use scenarios seem not to contribute to the improvement of the area in terms of its sustainable management, a wide range of alternative scenarios may be developed with the DIS model, also based on policies of particular relevance to the region and help to determine the potential desertification consequences of these policies in this spatially complex landscape.

# 9. CONCLUSIONS



## 9.1. MAIN RESEARCH CONCLUSIONS

Hilly Mediterranean environments, that are almost totally terraced and mostly abandoned such as the Serra de Rodes catchment, with steep slopes, shallow soils and low management, are subject to soil degradation or rehabilitation through complex biophysical processes further influenced by economic and social conditions (Dunjó et al., 2003).

Soil erosion has been considered as the main process of land degradation in the Serra de Rodes catchment, although the degradation of the vegetative cover as well as physical, chemical and biological soil-degradation processes are also believed to have an important role in the target area. In general, soil degradation is perceived as a major threat in the Mediterranean region due to changes in land use and possible future climate change (Cammeraat and Imeson, 1998).

### 9.1.1. LAND-DEGRADATION ASSESSMENT AT PLOT SCALE

It is well known that soils on stable landscape surfaces and under good plant cover conditions may improve with time by accumulating organic material, increasing floral and faunal activity, enhancing soil aggregate stability, increasing infiltration capacity and decreasing erosion (Trimble, 1990). In the case of matorral or shrub vegetation, the study of the effects of their characteristics indicates an appreciable control of erosion and runoff at both hillslope and catchment scale. Thus, it is postulated that a natural soil-restoration process may occur in well-vegetated environments, although its occurrence may be delayed because of adverse conditions related to soil condition at the time of agricultural set-aside and because of periodic wildfires (Dunjó et al., 2003).

Conversely, the results have shown that those cultivated fields such as vineyards and olive trees, with low soil fertility, are prone to enhance soil-degradation processes, either if inadequate or non-management practices are applied, or they are abandoned. For example, the development of surface seals in cultivated soils during rainstorms reduces infiltration rates, increases surface runoff and the erosion hazard, and consequently loss of organic matter and soil fertility. Many authors have demonstrated that in a wide range of environments both runoff and sediment loss decrease exponentially as the percentage of vegetation cover increases (Elwell and Stocking, 1976; Lee and Skogerboe, 1985; Francis and Thornes, 1990). In general, this pattern is also true when assessing the overall set of selected environments in the Serra de Rodes catchment, and therefore, one of the main conclusions that may be drawn from this study is the significant influence of land-use/cover on runoff and soil-erosion processes in the target area (e.g. Vacca et al., 2000; Dunjó et al., in press).

The principal components analysis (PCA) allowed the development of three simplified soil quality indicators capable of representing more complex physical and chemical data (Dunjó et al., 2003), being very useful to describe soils in terms of the land sustainability and management (Halvorson et al., 1997; Sims et al, 1997). The results have given evidence of the fragility of the assessed environments, especially the cultivated ones, in terms of runoff generation, sediment yield and nutrient losses, accounting for their vulnerability to land degradation processes. From the overall set of variables assessed in the PCA, the ones revealed as significant in the analysis such as:

rainfall amount, runoff generation, dissolved calcium, magnesium, sodium and potassium, dissolved organic carbon and nitrogen, soil erosion, eroded organic carbon and nitrogen, vegetal debris, aerial plant canopy, water holding capacity, particle size distribution and textural class, cation exchange capacity, soil organic matter, total nitrogen, bulk density, mechanic impedance, slope angle, undergrowth plant canopy, bare soil and stone cover, are considered the main factors of land degradation in the Serra de Rodes catchment.

### 9.1.2. DEVELOPMENT AND APPLICATION OF THE DIS MODEL

For the development of the DIS model at a catchment scale, soil erosion has been considered the main indicator of land degradation and desertification in the target area. In general, soil erosion represents a serious hazard resulting in land degradation and desertification in the Mediterranean region, bringing about large reductions in vegetation growth, siltation of water courses, filling of valleys and reservoirs, and the formation of deltas along coastal areas. In most Mediterranean lands, soil-erosion rates have been influenced by people since early prehistoric times (Inbar, 1992).

The general results seem to indicate that there are no important degradation processes in the natural vegetation of the Serra de Rodes field site and that after abandonment there is a recovery process through secondary succession, which might be considered as an indicator of the system's resilience (Dunjó et al., in press). It is important to state that this research has tackled soil erosion as the mobilisation of soil particles from upper to lower parts of the study catchment but not as the absolute loss of material, and therefore soil erosion and runoff rates registered in the experimental site, may be considered low (e.g. López-Bérmudez et al., 1998).

The main factors of land degradation identified at plot scale, such as the textural class and particle size distribution, bulk density, vegetation cover, slope angle, runoff and soil erosion have been applied as input variables in the DIS model, and so they seem to be also relevant at catchment scale. Moreover, an additional set of data such as monthly rainfall amount, soil-moisture content, and soil depth have been also required in the DIS development, although they were not significant in the PCA.

#### 9.1.2.1. Sensitivity analysis

Three scenarios relating to relatively dry, normal and wet rainfall years were assessed in order to determine whether soil erosion and/or runoff were sensitive to the amount of precipitation. The main results demonstrate that the DIS model is moderately sensitive to rainfall and in particular to low rainfall amounts such as in the dry scenario.

Due to the non-linearity of the DIS model, it is quite difficult to rank the sensitivity of the input parameters, although the significance of the soil erosion related parameters (SEP) and the hydrological parameters (HYP) to the overall sensitivity analysis is clear. The soil erodibility factor is revealed as the key input variable of the SEP, although the discharge-scaling factor as well as the soil depth are also important with regards to soil erosion. On the other hand, the saturated hydraulic conductivity is revealed as the most sensitive parameter of the overall set of input variables, for both runoff and erosion. Thus, the value of these variables in the model is extremely important for the outcomes and so their correct determination is crucial. Conversely, the vegetation cover and the slope constant are the most unresponsive factors from the SEP, as is the b parameter from the HYP, and therefore their role in the model results is not decisive.

#### 9.1.2.2. Validation analysis

The first conclusion that may be drawn from the validation analysis is that the use of plot-scale data from the runoff-erosion microplots is not the most appropriate for validating the DIS model, which was originally developed for the catchment scale. It seems clear from the results that there is an important scaling problem between the field and simulated data (Zhang, 1999).

The model overestimates soil erosion, especially in the cultivated environments, and it underestimates it in the most vegetated environments. Regarding overland flow, field and modelled data have the same order of magnitude, although they show very different patterns. The simulated runoff is overestimated and underestimated in the same land uses as the soil erosion.

However, in general, both datasets (measured and simulated) agree in determining the lowest erosion and runoff rates, in the plots remaining natural, with vegetation cover, while the highest rates are found in the land uses with cut vegetation, or crops. Thus, although data are not very reliable from a quantitative point of view, the modelled data are revealed to be significant in terms of which land uses are the most and least potentially degraded and so which scenarios would be better for preventing land degradation and which would accelerate land degradation. The DIS may need to be more complex, have better and more input data and a regional scale validation if the magnitudes were to be predicted reliably.

#### 9.1.2.3. Land-use scenario outcomes

Both land-use scenario outcomes, show greater soil-erosion and runoff rates than the ones modelled for the original landscape (S0). The cultivated scenario (S2) has been revealed as the most potentially erodible and more prone to overland flow. In S2, soil erosion is almost double than in S0 and in the wildfire scenario (S1), and rates are even greater when assessing runoff. Thus, a scenario mostly covered by cultivated fields, it is not likely to be a better management strategy in terms of preventing land degradation in the study area.

The determination of the level of soil degradation of each land-use scenario has been based on the FAO/UNEP/UNESCO (1979) classification, which accounts for the annual soil loss as an indicator of soil erosion by water. The three scenarios are under moderate land degradation (0.6-3.3 mm/yr), and on the basis of this classification, it seems that the simulated results in at a catchment scale are in the right order of magnitude for soil erosion values. This is a catchment scale verification of the model outcomes.

## 9.2. RESEARCH CONSTRAINTS

### 9.2.1. TIME

In the present research work, the most time consuming tasks have been on the one hand the field work and laboratory analysis which required more than a year of work at the beginning of the study research, and on the other hand, the model research and development which has required more than a year and a half to complete including its verification, sensitivity analysis, validation and the run of the scenarios outcomes.

### 9.2.2. FUNDING RESOURCES

The fieldwork and laboratory analysis have been supported by the CICYT: “Comisión Interministerial de Ciencia y Tecnología” of the Spanish Government (MECD), through a small part of the budget of the subproject: "The effect of land use and land use change in Mediterranean ecosystems" (1998-2000). Therefore, bearing in mind the available resources, the experimental design for this project was planned accordingly in terms of the size of the runoff-erosion microplots and their monitoring, as well as the field determinations and the associated chemical and physical analysis.

### 9.2.3. DATA AVAILABILITY

#### 9.2.3.1. Meteorological data

Meteorological data to play an essential role in the outcomes of the DIS model, since they are not only required for the development of the model, but also for the sensitivity analysis, the validation and the land use scenario outcomes. Thus, reliable, precise and accurate meteorological data from the Serra de Rodes catchment were highly desirable. The best option would have been to have data from an automatic weather station (AWS) ideally located in the catchment area, since this kind of equipment allows the continuous recording of meteorological data such as rainfall amount and intensity, solar radiation, humidity, and temperature. However, this was not available and the required data were provided by the Figueres AWS, located in the vicinity of the study area (approximately 20 km).

#### 9.2.3.2. Soil erosion and overland flow data

In order to perform the best validation of the DIS model, measured soil erosion and runoff data, would have been desirable at a catchment scale. However, field data were only available from the monitoring of runoff-erosion microplots of 1 $m^2$ area. Thus, the validation of the DIS had to be compatible with the available data at a one-cell scale, and the minimum resolution of the model (1 pixel = 25 $m^2$), overlooking the significant contribution of factors such as the digital elevation model (DEM) and local drainaige direction (LDD), on the generation of the overland flow and consequently on the sediment yield at a catchment scale. Likewise, adjustments were also made with regards to the time scale in the model validation, since field data were related to individual rainfall events and the timestep in the model is a month, because at a larger scale, meteorological data is only available at this resolution. As a result, it could be possible that the model validation outcomes are not as good as they could have been if a larger scale dataset had been available to perform the validation at the catchment scale.

## 9.3. FURTHER RESEARCH WORK

After having performed the verification, the sensitivity analysis and the validation assessment of the DIS, it is believed that by modifying some internal factors, as well as including new ones, the model outcomes could be substantially improved. The following suggestions aim to contribute to the enhancement of the DIS model results:

- In the DIS model, monthly rainfall (mm) is converted into hourly rainfall (mm) through the rainfall frequency distribution (RFD), in order to calculate better values of actual infiltration capacity and consequently also runoff water. However, due to the lack of available RFD from the study site, the one used in the model is related to an area located in the Penedés region, in the interior part of Catalunya. Therefore, although this area is also considered semi-arid, the rainfall regime in terms of intensities experienced may be quite different, leading to possible errors in the model outcomes. Hence, it would be interesting to find a RFD from the study site or an area closer to it, and run the model using with the new dataset in order to check whether the results present significant differences or not.

- With reference to the validation of the model, the rock fragment has been demonstrated to be potentially important factor in the study area, by presumably influencing the generation of runoff water and sediment yield, especially in the pine trees afforestation environment with 47% of the total surface covered by stones. Thus, since in general the soils of the Serra de Rodes catchment are characterised by a quite high rock fragment cover, it would be interesting to include this factor in the DIS model in order to check whether significant changes or not are reflected in the results.

- The selected soil-erosion model (Thornes, 1990), which incorporates the dominant physical controls of hydrology and vegetation cover, was selected for the DIS model for its simplicity and tested applicability to arid and semi-arid Mediterranean environments (e.g. Zhang, 1999). However, the validation shows that it overestimates erosion in several land uses and in particular in those less vegetated ones such as the cultivated olive trees (COT) and vineyards (CV). So, the results strongly suggest that the vegetation cover plays an important role in the soil erosion model when assessed at a plot scale. However, only the percentage of the vegetation cover is considered, omitting its type and structure, spatial pattern and a unique constant value per land use is used throughout the overall run of the DIS model.

Here, bearing in mind that the aim of the model is to be simple and use the least possible data, the incorporation of specific characteristics of the vegetation are not considered, since it seems likely that they would make the model too complex. Thus, it seems probable that the best and easiest way of obtaining more reliable vegetation data is to determine its percentage according to each season of the year, and apply them in the soil erosion model in a more dynamic way. The DIS would be run in order to assess whether significant differences exist in the soil erosion outcomes, when adding this dynamic component.

- Another factor that could contribute to the enhancement of the model performance is the sediment transport capacity equation, which has been tested in the validation results at a plot scale, and which substantially reduces net soil erosion. Thus, it is suggested the incorporation of this simple equation described by Kirkby (1976), to the DIS model script in order to assess differences between the original and this extended version of the model when running it at a catchment scale.

- Completely terraced landscapes such as the Serra de Rodes catchment are a common trait in Mediterranean environments. However, little research has been addressed to the study of the effects of these anthropic infrastructures on the hydrological behaviour of the soils. Thus, it is believed here that it would be very interesting to incorporate a new component in an extended version of the DIS model that would take into consideration the influence of these agricultural terraces on the runoff and soil erosion rates, always bearing in mind that the DIS has to be simple and easy to perform, not only from a mathematical point of view but also in terms of the availability of the required data.

- It is believed that since the model was developed for a catchment scale, a set of field data at this same scale would be more suitable for its validation and therefore better results than with plot scale data should be expected. Hence, the testing of the DIS with a new dataset from a catchment or area in the Mediterranean region, with similar climatic, soil and vegetation characteristics, could reflect a better performance of the model leading to the development of better and more reliable land use scenarios alternatives for the study area.

- Finally, on the basis of the performed sensitivity analysis, it would be recommendable to check whether the most sensitive parameters are determined in the best and most reliable way, since they are the most determinant factors for the DIS outcomes. In particular, the saturated hydraulic conductivity (SatHK), which has been revealed as the most sensitive variable in the DIS, has been estimated on the basis of the HYDROPAR spreadsheet model, which uses a pedo-transfer function (PTF). Alternatively, the SatHK could be measured in the field. Calculations based on measured data are always believed to be more precise and accurate in terms of obtaining a more reliable approach to the real world than data simulated by any model. Another important factor is the erodability factor (K), which could be determined on the basis of experimental work at a point scale, in order to determine the best value for the soils of the study area. However, the problem of scaling up these measurements to the region scale would have to be taken into account.

Bearing in mind the previously discussed limitations of the model, to a large extent, the DIS has accomplished the main aim of this research, in terms of providing an easy and reliable tool for identifying present and future threats of land degradation and to develop effective strategies for achieving a sustainable management of the Serra de Rodes catchment.

# REFERENCES

Abrahams, A.D. and A.J. Parsons (1994). Hydraulics of interrill overland flow on stone-covered desert surfaces. In: J.Poesen and H. Lavee (Eds.), Rock Fragments in Soil: Surface Dynamics. Catena, 23: 111-140.

Acton, D.F. and Gregorich, L.J. (1995). Understanding soil health. In: D.F. Acton and L.J. Gregorich (eds.). The Health of Our Soils: Toward Sustainable Agriculture in Canada. Centre for Land and Biological Resources Research, Research Branch, Agriculture and Agri-Food Canada, Ottawa, 5-10 pp.

Acton, D.F. and Gregorich, L.J. (1995). Understanding soil health. In: D.F. Acton and L.J. Gregorich (eds.). The Health of Our Soils: Toward Sustainable Agriculture in Canada. Centre for Land and Biological Resources Research, Research Branch, Agriculture and Agri-Food Canada, Ottawa, 5-10 pp.

Adriaanse, A. (1993). Environmental Policy Performance Indicators. SDV Publishers, The Hague.

Agro, K.E., Bradley, C.A., Mittmann, N. (1997). Sensitivity Analysis in Health Economic and Phrmacoeconomic Studies: an Appraisal of the Literature. Pharmacoeconomics, 11(1): 75-88.

Albareda, J.M. (1955). Influence des changements de la vegetation dans les sols arides. In: Plant Ecology, Arid Zone Research, UNESCO, Paris, 84-88 pp.

Alexander, M. (1980). Effect of acidity on microorganisms and microbial processes in soil. In: T. Hutchinson and M. Harvas (eds.). Effects of Acid Precipitation on Terrestrial Ecosystems. Plenum Publishing, New York, 363-364 pp.

Alhgrin, C. E. (1974). The Effet of Fire on Soil Organisms. A: Kozlowski, T. T. Et Ahlgrin, C. E. (eds.). Fire and Ecosystems. New York: Acad. Press.

Allen, R.G., Pereira, L.S., Raes, D., Smith, M. (1998). Crop evapotranspiration: guidelines for computing crop requirements. FAO irrigation and drainage paper 56. FAO- Food and Agriculture Organization of the United Nations, Rome.

Ambrust, D.V.(1987). Principles of soil erosion. Detachment, movement and deposition. Proc. Symp. on Soil Erosion by Wind, Denver Co., Chapter of Soil Cons. Soc. Am. And USDA-SCS, 13 p.

Ariens, G.A., Van Mechelen, W., Bongers, P.M., Bouter, L.M., van der Wal, G. (2000). Physical Risk Factors for Neck Pain. Scandinavian Journal of Work Environment & Health, 26(1): 7-19.

Aru, A. and G., Barrocu (1993). The Rio Santa Lucia catchment area. In: Mediterranean Desertification and Land Use, MEDALUS Final Report (unpublished). Commission of the European Communities. Contract number EPOC-CT900014-(SMA), 533-559 pp.

Aubreville, A. (1949). Climats, Forets et Desertification de l'Afrique Tropicale. Paris, Societe d'Editions Geographiques, Maritimes et Coloniales.

Bajaracharya, R.M. and R., Lal (1992). Seasonal soil loss and erodibility variations on a Miamian silt loam soil. Soil Science Society of American Journal, 56: 1560-1565.

Baker, S., D. Ponniah, Smith, S. (1999). Survey of Risk Management in Major UK Companies, Journal of Professional Issues in Engineering Education and Practice, 125(3): 94-102.

Bakkes, J.A., van den Born, G.J., Helder, J.C., Swart, R.J., Hope, C.W., Parker, J.D.E. (1994). An overview of environmental indicators: State of the Art and Perspectives. UNEP/ EATR. 94-01; RIVM/402001001. Environmental Assessment Sub-Programme; UNEP, Nairobi.

Balba, M.A. (1995). Management of problem soils in arid ecosystems. Boca Raton, FL: CRC Lewis Publishers. 179-239 pp.

Baldock, J.A., and P.N., Nelson (2000). Soil Organic Matter. In: Handbook of Soil Science (Sumner, M.E., ed.). CRC Press LLC, Boca Raton, Florida. B 25 - 27 pp.

Baniotopoulos, C.C. (1991). A Contribution to the Sensitivity Analysis of the Sea-Bed Structure Interaction Problem for Underwater Pipelines. Computers & Structures, 40(6): 1421-1427.

Barrow, C.J. (1994). Land Degradation: development and breakdown of terrestrial environments. Cambridge University Press.

Beck, M.B., Ravetz, J.R., Mulkey, L.A., Barnwell, T.O. (1997). On the Problem of Model Validation for Predictive Exposure Assessments. Stochastic Hydrology and Hydraulics, 11(3): 229-254.

Birkeland, P.W. (1999). Soils and Geomorphology (3rd edition). Oxford University Press.

Blake, G.R. and K.H. Hartge (1986). Bulk density. In: A. Klute (ed.). Methods of soil analysis. Part1. Physical and mineralogical methods. 2nd ed. Agronomy 9: 363-382.

Blöschl, G.; Grayson, R. (2001). Spatial Observations and Interpolation. In: Blöschl, G.; Grayson, R. (eds.), Spatial Patterns in Catchment Hydrology: observations and modelling. Cambridge University Press.

Bochet, E., Poesen, J., Rubio, J.L. (1995). Influence of three Mediterranean plants species on the spatial and temporal variation of water erosion (Valencia, Spain): experimental set up and preliminary results. In: Desertification in a European Context: Physical and Socio-economic Aspects (eds R. Fantechi et al.), pp. 377-388. Proceedings of the European School of Climatology and Natural Hazards Course. El Campello, Pueblo Acantilado, Alicante, 6-13 October, 1993. CEE, Luxembourg.

Bochet, E., Rubio, J.L., Poesen, J. (1998). Relative efficiency of three representative matorral species in reducing water erosion at the microscale in a semi-arid climate (Valencia, Spain). Geomorphology, 23: 139-150.

Boix-Fayos, C., Calvo-Cases, A., Imeson, A.C., Soriano-Soto, M.D., Tiemessen, I.R. (1998). Spatial and short-term temporal variations in runoff, soil aggregation and other soil properties along a mediterranean climatological gradient. Catena, 33: 123-138.

Bouma, N.A., and A.C. Imeson (2000). Investigation of relationships between measured field indicators and erosion processes on badland surfaces at Petrer, Spain. Catena, 40: 147-171.

Bradford, J.M. (1986). Penetrability. In: Methods of Soil Analysis. A. Klute, (2nd edition). Am. Soc. Agron. Madison, W1. 1188 p.

Bremner, J. M. and C. S. Mulvaney (1982). Nitrogen-total. p. 595-624. In: A. L. Page, et al. (ed.) Methods of soil analysis: Part two. Chemical and microbiological properties. ASA Monograph Number 9.

Brun, R, Reichert, P., Kunsch, H.R. (2001). Practical Identifiability Analysis of Large Environmental Simulation Models. Water Resources Research, 37(4):1015-1030.

Calkins, H.W. and Tomlinson, R.F. (1977). Geographic Information Systems: Methods and Equipment for Land Use Planning. Ottawa, International Geographical Union, Commission of Geographical Data Sensing and Processing and U.S. Geological Survey.

Camberato, J.J. (2000). Bioavailability of Calcium, Magnesium and Sulfur. In: Sumner, M.E. (ed.), Handbook of soil science. CRC Press, Boca Raton, Florida, D-53 – D-59 pp.

Cammeraat, L.H. and A.C. Imeson (1998). Deriving indicators of soil degradation from soil aggregation studies in southeastern Spain and southern France. Geomorphology, 23: 307-321.

Campbell, D.J. and Henshall, J.K. (1991). Bulk density. In: K.A. Smith and Ch.E. Mullis: Soil analysis. Marcel Dekker, Inc. 329-366 pp.

Campbell, G.S. (1985). Soil physics with basic: Transport models for soil-plant systems. Elsevier Science Publishers, The Netherlands, 40-97 pp.

Carlstein, T. (1982). Time, Resources, Society and Ecology: on the capacity for human interaction in space and time (vol. I. Pre-industrial societies). London: George Allen and Unwin.

Carter, D.L. (1969). Managing moderately saline (salty) irrigation water. Univ. ID Curr. Inf. Ser. 107.

Cerdà, A. (1996). Seasonal variability of infiltration rates under contrasting slope conditions in southeast Spain. Geoderma, 69: 217-232.

Cerdà, A., Imeson, A.C., Calvo, A. (1995). Fire and aspect induced differences on the erodability and hydrology of soils at La Costera, Valencia, southeast Spain. Catena, 24: 289-304.

Chapman, H.D. (1965). Cation exchange capacity. In: C.A. Black (ed.) Methods of soil analysis – Chemical and microbiological properties. Agronomy Monograph 9: 891-901.

Cheng, T.C.E. (1991). EPQ with Process Capability and Quality Assurance Considerations. Journal of the Operational Research Society, 42(8): 713-720.

Chepil, W.S. and N.P. Woodruff (1963). The physics of wind erosion and its control. Adv. Agron., 15:211-302.

Chevalier, S., Choiniere, R., Bernier, L. (1992). User Guide to 40 Community Health Indicators. Community Health Division., Health and Welfare Canada, Ottawa.

Clark, W.C. and Munn, R.E. (eds.) (1986). Sustainable development of the Biosphere. Cambridge: Cambridge University Press.

Conacher, A.J. and M. Sala (1998). Land degradation in the Mediterranean environments of the world: nature and extent, causes and solutions. John Wiley.

Cullen, A.C., and H.C. Frey (1999). Probabilistic Techniques in Exposure Assessment. Plenum Press: New York.

Darmody, R.G. and L.D. Norton (1994). Structural degradation of a prairie soil from long-term management. In: Ringrose-Voase, A.J; Humphreys, G.S. (eds.), Soil Micromorphology: Studies in Management and Genesis. Developments in Soil Science 22. Elsevier, Amsterdam, 641-650 pp.

DeBooysen, P. and Tainton, N.M. (eds.) (1984). Ecological Effects of Fire in S.African Ecosystems. Berlin: Springer-Verlag.

DESERTLINKS (2001). Internal preliminary report.

Dever, G.E.A. (1979). Community Health Analysis. Aspen Systems Corporation. Germantown, Marytown.

Dingman, S.L. (2002a). Physical Hydrology (2nd edition). Prentice-Hall, Inc., Upper Saddle River, New Jersey. 222-246 pp.

Dingman, S.L. (2002b). Physical Hydrology (2nd edition). Prentice Hall, Inc. Upper Saddle River, New Jersey, 25-35 pp.

Doerge, T. (2001). Fitting soil electrical conductivity measurements into the precision farming toolbox. Wisconsin Fertilizer, Aglime and Pest Management Conference. Madison, WI, January 16-18. In: http://www.soils.wisc.edu, accessed on February 2003.

Doran, J.W. and T.B. Parkin (1994). Defining and assessing soil quality. In: J.W. Doran, D.C. Coleman, D.F. Bezdicek, and B.A. Stewart (eds.). Defining Soil Quality for a sustainable Environment. SSSA Special Publication No. 35. Soil Science Society of America, Madison, WI, 3-21 pp.

Doran, J.W. and T.B. Parkin (1996). Quantitative indicators of soil quality: a minimum data set. In: J.W. Doran and A.J. Jones (eds.). Methods for assessing soil quality. Soil Science Society of America, Madison, WI, 25-38 pp.

Doran, J.W., Jones, A.J., Arshad, M.A., Gilley, J.E. (1999). Determinants of Soil Quality and Health. In: Lal, R. (ed.), Soil Quality and Soil Erosion. CRC Press.

Douglas, M. (1997). Guidelines for the Monitoring and Evaluation of Better Land Husbandry, Association for Better Land Husbandry, 28 p.

Dregne, H.E. (1976). Desertification: Symptoms of a crisis. In: Desertification, process, problems, perspectives. University of Arizona Press.

Dregne, H.E. (1982). Desertification in the Americas. Symposia Papers III, Desertification and Soils Policy, 12th I.S.S.S. Congress, New Delhi.

Dregne, H.E. (1983). Desertification of Arid Lands. Chur: Harwood Academic Publishers.

Dregne, H.E. (1986). Desertification of arid lands. In: Physics of desertification, (ed. F. El-Baz and M.H.Aa. Hassan). Dordrecht, The Netherlands: Martinus, Nijhoff.

Dregne, H.E. (1991). Human Activities and Soil Degradation. In: Skujins, J. (ed.), Semiarid lands and deserts: soil resource and reclamation, Marcel Dekker, Inc., New York.

Dunjó, G., Pardini, G., Gispert, M. (2003). Land use change effects on abandoned terraced soils in a Mediterranean catchment, NE Spain. Catena, 52: 23-37.

Dunjó, G., Pardini, G., Gispert, M. (in press). The role of land use-land cover on runoff generation and sediment yield at a microplot scale, in a small Mediterranean catchment. Journal of Arid Environments.

El-Hinnawi, E. and M. Hashmi (1982). Global Environmental Issues. United Nations Programme, Nairobi, Kenya.

Eltz, F.L. and L.D. Norton. (1997). Soil surface roughness changes as affected by tillage and rainfall erosivity. Soil Sci. Soc. Am. J.

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

ERDAS (1997). Field Guide Manual. Atlanta, Georgia: ERDAS, Inc.

ESRI (2000). ArcInfo ver. 8.0.2 manual (on-line documentation), Environmental Systems Research Institute, Redlands, CA.

FAO (1984). Provisional methodology for assessment and mapping of desertification. Food and Agriculture Organization of the United Nations and United Nations Environment Programme, Rome, Itlaly, 84 p.

FAO Constancinesco (1967). Soil Conservation for Developing Countries. Soils Bull., No. 30

FAO, Erosion by Water (1965). FAO Agric. Dev. Report No. 81, FAO, Rome.

FAO/UNEP/UNESCO (1979). Provisional Methodology of Soil Degradation Assessment, 73 p.

Fogh Mortensen, L. (1997). The driving force-state-response framework used by CSD. In: Moldan, B.; Billharz, S.; Matravers, R. (eds.), Sustainability Indicators: Report of the project on Indicators of Sustainable Development (SCOPE). John Wiley & Sons.

Font, J. (2000). Comunitats vegetals: entre el manteniment de la biodiversitat i l'amenaça del foc. In: Revista de Girona. Cap de Creus, espai natural, espai cultural: Valors ecològics, paisatgístics i literaris del vertex més oriental de la Penínsul Ibèrica. Edició: Diputació de Girona, 198: 70-72 pp.

Fortià, R. (1993). El Medi Natural de les comarques gironines. L'estat de la qüestió. Diputació de Girona, 138-153 pp.

Fotheringham, A. S. and M. Wegener (eds.) (1999). Spatial models and GIS: new potential and new models. Taylor and Francis, New York.

Fraedrich, D. and A. Goldberg (2000). A Methodological Framework for the Validation of Predictive Simulations. European Journal of Operational Research, 124(1): 55-62.

Francis, C.F. and J.B., Thornes (1990). Runoff hydrographs from three Mediterranean vegetation cover types. In: J.B. Thornes (ed.), Vegetation and Erosion, Processes and Environments. Wiley, Chichester, 363-384 pp.

Franquesa, T. (1995). El paisatge vegetal de la peninsula del Cap de Creus. Institut d'Estudis Catalans: Arxiu de les seccions de ciències, CIX secció de les ciències biològiques (Ed.), Barcelona.

Frey, H.C. and S.R. Patil (2001). Proceedings of the workshop on Sensitivity Analysis Methods at the Engineering Graduate research Center on NC State's Centennial Campus, June 11-12, 2001. http://www.ce.ncsu.edu/risk/pdf/frey.pdf, accessed on April 2003.

Fryberger, S.D. (1979). Dune forms and wind regime. In: McKee, E.D. (ed.), A study of global sand seas, pp.136, U.S. Geological Survey Professional Paper 1052.

Frye, W.W.; Ebelhar, S.A.; Murdock, L.W.; Blevins, R.L. (1982). Soil erosion effects on properties and productivity of two Kentucky soils. Soil Sci. Soc. Am. J. 46: 1051-1055.

Fryreer, D.W. (1987). Principles of soil erosion control: Cultivated lands. Proc. Symp. Soil Erosion by Wind. Soil Cons. Soc. Am. And USDA-SCS, 16 p.

Gallart, F. and P. Llorens (2002). Water resources and environmental change in Spain. A key issue for sustainable integrated catchment management. In: J.M. García-Ruíz, J.A.A. Jones and J. Arnáez (eds.), Environmental change and water sustainability. Instituto Pirenaico de Ecología, Consejo Superior de Investigaciones Científicas, Zaragoza, 11-20 pp.

Gallopin, G.C. (1996). Environmental and sustainability indicators and the concept of situational indicators. A system approach. Environmental Modelling & Assessment 1:101-117.

Gallopin, G.C. (1997). Indicators and their use: Information for Decision-making. In: Moldan, B.; Billharz, S.; Matravers, R. (eds.), Sustainability Indicators: Report on the Project on Indicators of Sustainable Development (SCOPE). John Wiley and Sons.

García-Ruíz, J.M; Lasanta, T.; Ortigosa, L.; Ruíz-Flaño, P.; Mart, C., González, C. (1995). Sediment yield under different land uses in the Spanish Pyrenees. Mountain Research and Development, 15 (3): 229-240.

García-Ruíz, J.M; Ruíz-Flaño, P.; Lasanta, T.; Montserrat, G.; Martínez-Rica, J.P., Pardini, G. (1991). Erosion in abandoned fields. What is the problem? In: Soil erosion studies in Spain (Sala, M.; Rubio, J.L. and J.M. García-Ruíz, eds.). Geoforma Ediciones, Logroño, 97-108 pp.

Gardiner, R. (2002). Towards Earth Summit 2002 Project. Stakeholder Forum for Our Common Future, London. In: http://www.earthsummit2002.org/Es2002.PDF, accessed on January 2003.

Garduno, M.A. (1977). Technology and desertification. UNCOD A/conf 74/6, Nairobi.

Glantz, M.H. (1977). Desertification: Environmental Degradation in and around Arid Lands. Boulder, Westview Press.

Glantz, M.H. and N.S. Orlovsky (1983). Desertification: A review of the concept. Desertification Control Bulletin 9: 15-22 pp.

Goldstein, J. (1977). Sensible sewage. Rodale Press, Emmaus, Pennsylvania

Goudie, A. S. (1990). Techniques for desert reclamation. John Wiley & Sons, Chichester.

Goudie, A.S. (1981). The Human impact: man's role in environmental change. Oxford: Basil Blackwell.

Halvorson, J.J., Smith, J.L., Papendick, R.I. (1997). Issues of scale for evaluating soil quality. J. Soil Water Conservation, 52: 26-30.

Hammond, A.; Adriaanse, A.; Rodenburg, E.; Bryant, D.; Woodward, R. (1995). Environmental Indicators: A systematic approach to measuring and reporting on environmental policy performance in the context of sustainable development. World Resources Institute, Washington, D.C.

Hardisty, J.; Taylor, D.M., Metcalfe, S.E. (eds.) (1993). Computerised Environmental Modelling: A Practical Introduction Using Excel. John Wiley & Sons Ltd., Chichester. 108-115 pp.

Hare, F.K. (1985). Climate variations, drought and desertification. WMO 635:35.

Hare, K. (1977). Connections between climate and desertification, Environmental Conservation 4(2): 82.

Hargreaves, G.H. and Z.A. Samani (1985). Reference crop evapotranspiration from temperature. Applied Engr. In Agric., 1 (2): 96-99.

Harris, R.F., Karlen, D.L., Mulla, D.J. (1996). A conceptual framework for assessment and management of soil quality and health. In: Doran, J.W.; Jones, A.J. (eds.), Methods for assessing soil quality SSSA Special Publication, 61-82 pp.

Helton, J.C.; and R.J. Breeding (1993). Evaluation of Reactor Accident Safety Goals. Reliability Engineering & System Safety, 39(2): 129-158.

Història dels Països Catalans (1992).

Holling, C.S. (1978). Adaptive Environmental Assessment and Management. John Wiley & Sons, Chichester.

Hopper, D. (1988). The World Bank's challenge: balancing economic need with environmental protection, Seventh Annual World Conservation Lecture, 3rd March, WWF United Kingdom, Godaiming.

Horton, R.E. (1940). An approach towards a physical interpretation of infiltration capacity. Proceedings of the Soil Science Society of America, 5: 399-417.

Hudson, N. (1971). Soil conservation. Batsford Ltd, London, 330 p.

Hutchinson, M.F. (1988). Calculation of hydrologically sound digital elevation models. Third International Symposium on Spatial Data Handling, Sydney. Columbus, Ohio: International Geographical Union.

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.

Hutnik, R. and Davis, G. (eds.) (1973). Ecology and reclamation of degraded land, vols. I and II. New York: Gordon and Breach.

Inbar, M. (1992). Rates of fluvial erosion in basins with a Mediterranean type climate. Catena, 19: 393-409.

Jacobs, J.M. and S.R. Satti (2001). Evaluation of reference evapotranspiration methodologies and AFSIRS crop water use simulation model. Unpublished final technical report, Department of Civil and Coastal Engineering, University of Florida, Gainesville, Florida.

Jakeman, A.J., Beck, M.B., McAleer, M.J. (eds.) (1993). Modelling change in environmental systems. Wiley, Chichester.

Kassas, M. (1977). Arid and semi-arid lands: Problems and prospects. Agro-Ecosystems 3:186.

Kates, R.W.,Johnson, D.L., Johnson Haring, K. (1977). Population, society and desertification. Report of UNCOD A/CONF 74/8, Nairobi.

Kay, B.D. and D.A. Angers (2000). Soil Structure. In: M.E. Sumner (ed.). Handbook of Soil Science. CRC Prees, Boca Raton, Florida. A-240- A-247 pp.

Kennedy, A.C.; Papendick, R.I. (1995). Microbial characteristics of soil quality. J. Soil Water Conservation. 50: 243-247

Kewley, R.H., Embrechts, M.J., Breneman, C. (2000). Data Strip Mining for the Virtual Design of Pharmaceuticals with Neural Networks. IEEE Transactions on Neural Networks, 11(3): 668-679.

Kilmer, V.J. and L.T. Alexander (1949). Methods of making mechanical analyses of soils. Soil Science 68: 15-24.

Kimaro, T., Tachikawa, Y., Takara, K. (2002). Groundwater Modelling Coupled with SVAT Model and its Application to the Yasu River Basin. Annuals of Disas. Prev. Res. Inst., Kyoto Univ., 45(B2), 163-172 pp.

Kirkby, M. and C. Kosmas (1999). Introduction. In: Kosmas, C.; Kirkby, M.; Geeson, N. (eds.), The Medalus project: Mediterranean desertification and land use. Manual on key indicators of desertification and mapping environmentally sensitive areas to desertification. Project report. European Commission.

Kirkby, M.J. (1976). Hydrological Slope Models: The Influence of Climate. In: E.Derbyshire (ed.), Geomorphology and Climate. Wiley, London. 247-267 pp.

Kirkby, M.J., Naden, P.S., Burt, T.P., Butcher, D.P. (eds.) (1993). Computer simulation in physical geography (2nd edition). John Wiley & Sons, Chichester.

Kjeldahl, J. (1883). A new method for the determination of nitrogen in organic matter. Z. Anal. Chem. 22: 366.

Kleijnen, J.P.C. (1995). Verification and Validation of Simulation-Models. European Journal of Operational Research, 82(1):145-162.

Kleijnen, J.P.C. and R.G. Sargent (2000). A Methodology for Fitting and Validating Metamodels in Simulation. European Journal of Operational Research, 120(1):14-29.

Klute, A. (1986). Water retention: laboratory methods. In: A. Klute (ed.) Methods of soil analysis: Part 1. Physical and mineralogical methods. ASA Monograph Number 9, 635-662 pp.

Kolka, R.K. and A.T. Wolf (1998). Estimating Actual Evapotranspiration for Forestal Sites: Modifications of the Thornthwaite Model. United States Department of Agriculture (USDA), Forest Service, Southern Research Station. Research Note SRS6, 1-8 pp.

Kosmas, C. (1998). Quantitative indicators of desertification. In: http://www.desertification.it/ASINARA%20web/11kosmas.htm, accessed on October 2001.

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.

Kosmas, C.; Ferrara, A.; Briassouli, H.; Imeson, A. (1999). Methodology for mapping environmentally sensitive areas (ESAs) to desertification. In: Kosmas, C.; Kirkby, M.; Geeson, N. (eds.), The Medalus project: Mediterranean desertification and land use. Manual on key indicators of desertification and mapping environmentally sensitive areas to desertification. Project report. European Commission

Kosmas, C.; Poesen, J., Briassouli, H. (1999). Key indicators of desertification at the Environmentally Sensitive Areas (ESA) scale. In: Kosmas, C.; Kirkby, M.; Geeson, N. (eds.), The Medalus project: Mediterranean desertification and land use. Manual on key indicators of desertification and mapping environmentally sensitive areas to desertification. Project report. European Commission.

Kosmas, C.S., Moustakas, N., Danalatos, N.G., Yassoglou, N. (1995). The effect of land use change on soil properties and erosion along a catena. In: J. Thornes and J. Brandt (eds.), Mediterranean Desertification and Land Use. John Wiley and Sons, Chichester, 207-227 pp.

Kovda, V. (1980). Land aridization and drought control. Boulder Westview Press.

Lal, R. (1988b). Erodibility and Erosivity. In: Soil Erosion Research Methods (ed R. Lal), Soil and Water Conservation Society, Ankeny, IA, 141 p.

Lal, R. (1988a). Soil degradation and the future of agriculture in sub-Saharan Agrica. J. Soil Water Conservation. 43: 444-451.

Lal, R. (1990). Water erosion and conservation: an assessment of the water erosion problem and the techniques available for soil conservation, chapter 6, 161-198 pp.

Lal, R. (1997). Degradation and resilience of soils. Philos. Trans. R. Soc. London Ser. B 352: 997-1010.

Lal, R. (1998). Soil quality and sustainability. In: Lal, R., Blum, W.H., Valentine, C., Stewart, B.A. (eds.). Methods for assessment of soil degradation. Advances in soil science. Boca Raton, FL, CRC Press, 17-35 pp.

Lal, R. (1999). Soil Quality and Food Security: The Global Perspective. In: Lal, R. (ed.), Soil Quality and Soil Erosion, CRC Press.

Lal, R. and B.A. Stewart (1990). Soil degradation. A global threat. Adv. Soil Sci. 11: xiii-xvii.

Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A. (eds.) (1997a). Soil Processes and the Carbon Cycle. CRC Press, Boca Raton, FL.

Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A. (eds.) (1997b). Management of Carbon Sequestration in Soil. CRC Press, Boca Raton, FL.

Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A. (eds.) (1995a). Soils and Global Change, Lewis Publishers, Boca Raton, FL, 440 p.

Lal, R., Kimble, J.M., Follett, R.F., Stewart, B.A. (eds.) (1995b). Soil management for mitigating the Greenhouse Effect. Lewis Publishers, Boca Raton, FL,.385 p.

Lal, R., Gall, G.F., Miller, F.P. (1989). Soil Degradation: I. Basic Processes. Land Degradation and Rehabilitation, Vol. 1, 51-69 pp.

Landres, P.B. (1992). Ecological indicators: panacea or liability. In: McKenzie, D.H.; McDonald, V.J. (eds.), Ecological indicators. Elsevier Applied Science, London, 1295-1318 pp.

Langaas, S. (1997). The Spatial dimension of Indicators of Sustainable Development: the role of Geographic Information Systems (GIS) and Cartography. In: Moldan, B.; Billharz, S.; Matravers, R. (eds.), Sustainability Indicators: Report of the project on Indicators of Sustainable Development (SCOPE). John Wiley & Sons.

Larson, W.E. and F.J., Pierce (1991). Conservation and enhancement of soil quality. In: Evaluation for Sustainable land management in the developing world, Vol.2. IBSRAM Proc. No.12 (2). International Board for Soil Research and Management, Bangkok, Thailand, 175-203 pp.

Lasanta, T. (1989). Evolución reciente de la agriculture de montaña: El Pirineo aragonés. Geoforma Ediciones, Logroño, 220 p.

Lasanta, T., García-Ruíz, J.M., Pérez-Rontomé, C., Sancho-Marcén, C. (2000). Runoff and sediment yield in a semi-arid environment: the effect of land management after farmland abandonment. Catena, 38: 265-278.

Le Bissonais, Y. (1990). Experimental study and modelling of soil surface crusting processes. In: Bryan, R.B. (ed.), Soil Erosion: Experiments and Models. Catena Verlag, Cremlingen-Destedt, 13-28 pp.

Le Houerou, H.N. (1975). The nature and causes of desertification. In: Proceedings of the IGU Meeting on Desertification, Cambridge, 22-26 September 1975.

Lean, G. (1995). Down to Earth: A simplified guide to the Convention to Combat Desertification why it is necessary and what is important and different about it. Secretariat of the United Nations Convention to Combat Desertification (UNCCD). Bonn, Germany.

Lee, C.R. and J.G., Skogerboe (1985). Quantification of erosion control by vegetation on problem soils. In: Al Swaify, W.C. Moldenhauer and A. Lo (eds.), Soil Erosion and Convervation. Soil Conservation Society of America, 437-444 pp.

Limat, S., Woronoff-Lemsi, MlC., Deconinck, E., Racadot, E., Jacquet, M., Herve, P., Cahn, J.Y. (2000). Cost-Effectiveness of Cd34(+) Dose in Peripheral Blood Progenitor Cell Transplantation for Non-Hodgkin's Lymphoma Patients: a Single Center Study. Bone Marrow Transplantation, 25(9): 997-1002.

López-Bermúdez, F., Romero-Díaz, A., Martínez-Fernandez, J., Martínez-Fernandez, J. (1998). Vegetation and soil erosion under a semi-arid Mediterranean climate: a case study from Murcia (Spain). Geomorphology, 24: 51-58.

Luk, S.H. (1979). Effect of soil properties on erosion by wash and splash. Earth Surface Processes, 4: 241-255.

Mabbut, J.A. (1978). The impact of desertification as revealed by mapping. Environmental Conservation 5-1: 45.

Maidment, D. R (1993). GIs and Hydrologic Modeling. In: Environmental Modeling with GIS. M.F. Goodchild, B.O. Parks, L.T. Steyaert (eds). Oxford University Press, New York, 147-167 pp.

Mainguet, M. (1994). Desertification: Natural background and human mismanagement, 2nd Edition. Springer-Verlag Berlin Heidelberg, Germany.

Mainguet, M. (1999). Aridity: droughts and human development. Spronger-Verlag, Berlin, Heidelberg, Germany, 249 p.

Manheim, L.M. (1998). Health Services Research Clinical Trials: Issues in the Evaluation of Economic Costs and Benefits. Controlled Clinical Trials, 19(2): 149-158.

Marcus, S. (1983). Towards a Semiotic Approach to Social Indicators. Paper presented at the International Congress of Logic, Methodology and Philosophy of Science, Salzburg, Austria, July 11-16, 1983.

Martínez Beltran, J. (1978). Drainage and reclamation of salt affected soils in the Bardenas area, Spain. International Institute for Land Reclamation and Improvement Publications No. 24, Wageningen, The Netherlands, 321 p.

Martínez-Fernández, J., López-Bermúdez, F., Martínez-Fernández, J. and Romero-Díaz, A. (1995). Land use and soil-vegetation relationships in a Mediterranean ecosystem: El Ardal, Murcia, Spain. Catena, 25: 153-167.

Martínez-Mena, M., Albaladejo, J., Castillo, V.M. (1998). Factors influencing surface runoff generation in a Mediterranean semi-arid environment: Chicamo watershed, SE Spain. Hydrological Processes, 12: 741-754.

McLaren, R.G.; Cameron, K.C. (1996). Soil Science: sustainable production and environmental protection. Oxford University Press, 3 p.

McQueen, D. and H. Noak (1988). Health Promotion Indicators: Current status, issues and problems. Health Promotion 3, 117-125 pp.

Meckelein, W. (1980). The problem of desertification within deserts. In: Desertification in extremely arid environments, IGU Working Group on Desertification in around Arid Lands, 24th International Geographical Congress, Tokyo.

MedAction (2001): Policies for land use to combat desertification. Research Project, European Commission. In: http://www.icis.unimaas.nl/medaction/downs/Brochure.doc, accessed on November 2002.

Mermut, A.R., Luk, S.H., Römkens, M.J.M., Poesen, J.W.A. (1997). Soil loss by splash and wash during rainfall from two loess soils. Geoderma, 75: 203-214.

Merz, J.F., Small, M.J., Fischbeck, P.S. (1992). Measuring Decision Sensitivity: A Combined Monte Carlo- Logistic Regression Approach. Medical Decision Making, 12(3): 189-196.

Middleton, H.E. (1990). Wind erosion and dust storm control. Chapter 3. In: Goudie, A.S. (ed.), Techniques for desert reclamation. John Wiley & Sons.

Ministerio de Agricultura (1986). Métodos Oficiales de análisis de suelos, III. Ministerio de Agricultura, Madrid, 166 p.

Ministerio de Medio Ambiente (2000). Internal report: "Informe sobre el programa de acción nacional contra la desertificación". Ministerio de Medio Ambiente, Secretaría General de Medio Ambiente, Dirección General de Conservación de la Naturaleza, España.

Monteith, J.L. (1965). Evaporation and the environment. Symposium of the Society of Exploratory Biology, 19: 205-234.

Morgan, M.G.; and M.Henrion (1990). Uncertainty: A Guide to Dealing With Uncertainty in Quantitative Risk and Policy Analysis. Cambridge University Press: Cambridge, N.Y.

Morgan, R.P.C. (1991). Soil Erosion and Conservation, Longman, Essex, 298 p.

Mulligan, M. (1996). Modelling Hydrology and Vegetation Change in a Degraded Semi-Arid Environment. Unpublished Thesis. University of London.

Mulligan, M. (1998). Modelling the geomorphological impact of climatic variability and extreme events in a semi-arid environment. Geomorphology, 24: 59-78.

Naveh, Z. (1990). Fire in the Mediterranean – a landscape ecological perspective. In: J.G. Goldammer and M.J. Jenkins (eds.), Fire in Ecosystems Dynamics. Mediterranean and Norhtern Perspective. SPB Academic Publishing, The Hague, 1-20 pp.

Nelson, D. W. and L. E. Sommers (1982). Total carbon, organic carbon and organic matter. In: A. L. Page et al. (ed.) Methods of soil analysis: Part 2. Chemical and microbiological properties. ASA Monograph Number 9, 539-579 pp.

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.

Norton, D., Shainberg, I. Cihacek, L., and J.H., Edwards (1999). Erosion and soil chemical properties. In: Lal, R. (Ed.), Soil Quality and Soil Erosion. CRC Press, Boca Raton, Florida, 39-56 pp.

Norton, L.D. (1987). Micromorphological study of surface seals developed under simulated rainfall. Geoderma 40: 127-140.

Oades, J.M. (1984). Soil organic matter and structural stability: Mechanisms and implications for management. Plant and Soil, 76: 319-337.

OECD (1993). Organization for Economic Cooperation and Development Core Set of Indicators for Environmental Performance Reviews. A Synthesis Report by the Group on the State of the Environment. OECD, Paris.

Oh, B.H. and L.H. Yang (2000). Sensitivity Analysis of Time-Dependent Behaviour in PSC Box Girder Bridges. Journal of Structural Engineering-ASCE, 126(2): 171-179.

Or, D. and J. Wraith (2000). Soil Water Content and Water Potential Relationships (Chapter 3). In: Handbook of Soil Science (Sumner, M.E., ed.). CRC Press LLC, Boca Raton, Florida. A-53-A-55 pp.

Ott, W.R. (1978). Environmental Indices: Theory and Practice. Ann Arbor Science, Michigan.

Paniagua, A., Kammerbauer, J., Avedillo, M. and A.M., Andrews (1999). Relationship of soil characteristics to vegetation successions on a sequence of degraded and rehabilitated soils in Honduras. Agriculture, Ecosystems and Environment, 72: 215-225.

Pardini, G., Dunjó, G., Gispert, M., Barrena, R., and G. Vigna Guidi (2002). Land use effects on soil response to runoff generation and sediment yield in the Serra de Rodes catchment, Alt Empordà, NE Spain. In: Rubio, J.L., Morgan, R.P.C., Ansis, S., Andreu, V. (eds.). Man and Soil at the Third Millennium. Geoforma Ediciones, Logroño, Vol. 2, 1323-1341 pp.

Pardini, G., Gispert, M. and G. Dunjó (2003). Runoff erosion and nutrient depletion in five Mediterranean soils of NE Spain under different land use. The Science of the Total Environment, 309: 213-224.

Parr, J.F.; Papendick, R.I.; Hornick, S.B.; Meyer, R.E. (1992). Soil quality: attributes and relationship to alternative and sustainable agriculture. Am. J. Altern. Agric. 7:2-3.

Pearce, F. (1986). Acid Rain. Harmondsworth: Penguin.

Pérez-Trejo, F. (1994). Desertification and land degradation in the European Mediterranean. Report EUR 14850 EN, European Commission.

Peuquet, D.; Davis, J.R. and S. Cuddy (1993). Geographic Information Systems and Environmental Modelling. In: Jakeman, A.J.; Beck, M.B.; McAleer, M.J. (eds.), Modelling Change in Environmental Systems. John Wiley & Sons.

Phillips, A.; Janies, D.; and W. Wheeler (2000). Multiple Sequence Alignment in Phylogenetic Analysis. Molecular Phylogenetics and Evolution, 16(3): 317-330.

Pimentel, D.; Terhune, E.C.; Dyson-Hudson, Rochereau, S.; Samis, R.; Smith, E.A.; Denman, D.; Reifschneider, D.; Shepard, M. (1976). Land degradation: effects on food and energy resources . Science, 194:145-155

Plujà, A. (2000). El Cap de Creus 60 itineraris de Portbou a Roses, 148-150 pp.

Poesen, J. (1995). Soil erosion in Mediterranean environments. In: Desertification in an European context: Physical and socio-economic aspects, Fantechi, R.; Peter, D.; Balabanis, P. and Rubio, J.L. (eds.). Proceedings of the European School of Climatology and Natural Hazards Course, Alicante, Spain. European Commission, Final Report.

Poesen, J. and Govers, G. (1990). Gully erosion in the loam belt of Belgium: Typology and control measures. In: Boardman, J. et al. (eds.), Soil erosion on agricultural land, John Wiley, Chichester, 513-530 pp.

Poesen, J. and H. Lavee (1994). Rock fragments in top soils: significance and processes. Catena, 23: 1-28.

Poesen, J. and Nearing, M. (1993). Soil surface sealing and crusting. Catena Supplement 24, Catena Verlag, Germany, 139 p.

Poesen, J., Ingelmo-Sanchez, F., Muncher, H. (1990). The hydrological response of soil surfaces of rainfall as affected by cover and position of rock fragments in the top layer. Earth Surface Processes and Landforms, 15: 653-671.

Poesen, J.W., Torri, D., Bunte, K. (1994). Effects of rock fragments on soil erosion by water at different spatial scales: a review. Catena, 23: 141-166.

Pons, P. (2000). Incendis, fauna i gestió del patrimoni natural. En: Revista de Girona. Cap de Creus, espai natural, espai cultural: Valors ecològics, paisatgístics i literaris del vèrtex més oriental de la península ibèrica. Edició: Diputació de Girona, 198:73-75 pp.

Porta, J., López-Acevedo, M. And C. Roquero (1994a).Edafología para la agricultura y el medio ambiente. Ediciones Mundi-Prensa, 264-265 pp.

Porta, J., López-Acevedo, M. And C. Roquero (1994b). Edafología para la agricultura y el medio ambiente. Ediciones Munidi-Prensa, Madrid, 226 p.

Priestley, C.H.B. and R.J. Taylor (1972). On the assessment of surface heat flux and evaporaion using large-scale parameters. Mon. Weat. Rev. 100: 81-82.

Puigdefàbregas, J. (1998). Implications of regional scale policies on land condition and land degradation in the Mediterranean basin. In: Global Change and Terrestrial

Ecosystems (Ed.), The Earth Changing Land. GCTE-LUCC Open Science Conference on Global Change. Barcelona: GCTE.

Quiroga, A. R., Buschiazzo, D. E., Peinemann, N. (1998). Management discriminant properties in semiarid soils. Soil Science, 163: 591-597.

Rana, G. (1998). A model of the Priestley-Taylor ? coefficient for evapotranspiration estimates: the reference crop case in a Mediterranean region. Agr. Med. 128: 11-20.

Rapp, A. (1974). A review of desertization in Africa- Water, Vegetation and Man. SIES Report 1. Secretariat for International Ecology, Stockholm, Sweden, 77 p.

Reining, P. (compiler) (1978). A Handbook on Desertification Indicators: based on the Science Associations' Nairobi Seminar on Desertification, Washington, American Association for the Advancement of Science.

Rhoades, J.D. (1982). Soluble salts. In: A.L. Page et al. (ed.) Methods of soil analysis: Part two: Chemical and microbiological properties. Monograph Number 9 (Second Edition). ASA, Madison, WI, 167-179 pp.

Romero-Díaz, A., Cammeraat, L.H., Vacca, A. and C. Kosmas (1999). Soil erosion at three experimental sites in the Mediterranean. Earth Surface Processes and Landforms, 24, 1243-1256.

ROSELT (Long term ecological monitoring observatories network) (1997). ROSELT: A contribution to development through its products. OSS (Sahara and Sahel observatory). In: http://firewall.unesco.org/oss/public/pubrosla.htm, accessed on December 2002.

Ruíz-Flaño, P. (1993). Procesos de erosion en campos abandonados del Pirineo. Geoforma Ediciones, Logroño, 9-19 pp.

Rutherford, I. (1997). Use of models to link indicators of sustainable development. In: Moldan, B.; Billharz, S.; Matravers, R. (eds.), Sustainability Indicators: Report of the project on Indicators of Sustainable Development (SCOPE). John Wiley & Sons.

Sabadell, J.E., Risley, E.M., Torgenson, H.T., Thornton, B.S. (1982). Desertification in the United States, Status and Issues, Bureau of Land Management, Department of the Interior.

Sala, M. and A. Calvo (1990). Response of four different Mediterranean vegetation types to runoff and erosion. In: Thornes, J.B. (ed.), Vegetation and Erosion. Wiley, New York, 343-362 pp.

Salehi, F.; Prasher, S.O.; Amin, S.; Madani, A.; Jebelli, S.J.; Ramaswamy, H.S.; and C.T. Drury (2000). Prediction of Annual Nitrate-N Losses in Drain Outflows with Artificial Neural Networks. Transactions of the ASAE, 43(5): 1137-1143.

Salinity Laboratory Staff (1954). Diagnosis and improvement of saline and alkali soils, U.S. Department of Agriculture Handbook Nº60. L.A. Richards (ed.)

Saltelli, A.; Chan, K.; and E.M. Scott. Eds (2000). Sensitivity Analysis. John Wiley and Sons, Ltd.: West Sussex, England.

Salvucci, G.D. and D. Entekhabi (1994). Explicit expressions for Green-Ampt (Delta function diffusivity) infiltration rate and cumulative storage. Water Resources Research, 30: 2661-2663.

Schollenberger, C.J. and R.H., Simon (1945). Determination of exchange capacity and exchangeable bases in soil-ammonium acetate method. Soil Science 59: 13-24.

Science Council of Canada (1986). A growing concern: Soil degradation in Canada. Council Statement, September 1986.

Seybold C.A., Herrick, J.E., Brejda, J.J. (1999). Soil resilience: a fundamental component of soil quality. Soil Science, 164: 224-234.

Sikora, L.J., Yakovchenko, V., Cambardella, C.A., Doran, J.W. (1996). Assessing soil quality by testing organic matter, In: Soil Science Society of America (Ed.), Soil Organic Matter: Analysis and Interpretation, Special Publication Nº46, Madison, WI, 41-50 pp.

Sims, J.T. (2000). Soil Fertility Evaluation. In: Sumner, M.E. (Ed.), Handbook of soil science. CRC Press, Boca Raton, Florida, D-113 – D-127 pp.

Sims, J.T., Cunningham, S.D., Sumner, M.E. (1997). Assessing soil quality for environmental purposes: Roles and challenges for soil scientists. J. Environmental Quality, 26: 20-25.

Singer, M.J.; Munn, D.N. (1987). Soils. Macmillan, New York, 492 p.

Soil Conservation Service (SCS) (1963). Research in the Great Plains States. Misc. Pub., 902.

Soil Science Society of America (1997). Glossary of Soil Science Terms. 1996, SSSA, Madison, WI, 134 p.

Soil Survey Division Staff (1993). Soil Survey Manual. Soil Conservation Service (SCS). U. S. Department of Agriculture (USDA), Handbook N°18. Washington D.C., U.S., Government Printing Office.

Soil Survey Laboratory Staff (1996). Soil Survey Laboratory Methods Manual. U.S. Department of Agriculture (USDA). Soil Conservation Service (SCS). Soil Survey Investigations Report No. 42, Version 3.0. Washington, D.C.

Soil Survey Staff (1996). Keys to soil taxonomy. US Government Printing Office, Washington, DC.

Soil Survey Staff (SSS) (1951). Soil Survey Manual, Bureau of Plant Industry, Soils and Agric. Eng. USDA. Handbook 18:251.

Soil Survey Staff (SSS) (1975). Soil Taxonomy: a basic system of soil classification for making and interpreting soil surveys. USDA Agricultural Handbook 436. U.S. Government Printing Office, Washington, DC, p. 754.

Spooner, B. and H.S., Mann (eds.) (1982). Desertification and Development: dryland ecology in social perspective. London: Academic Press.

Sposito, G. (2000). Ion Exchange Phenomena. In: Sumner, M.E. (Ed.), Handbook of soil science. CRC Press, Boca Raton, Florida, B-241 – B- 253 pp.

Steinke, A., Green, D.G., Peters, D.J. (1996). On-line environmental and geographic information systems. In: Saarenmaa, H. and Kempf, A. (eds.). Internet applications and electronic information resources in forestry and environmental sciences. EFI Proceedings 10: 89-98, Joensuu.

Stocking, M.A. and N. Murnaghan (2001). Handbook for the field assessment of land degradation. Earthscan Publication Ltd., London, UK.

Street, P. (1987). Land use policy. Butterworth, London, 4-4: 362.

Sumner, M. E. and W. P. Miller (1996). Cation exchange capacity , and exchange coefficients. In: D. L. Sparks (ed.) Methods of soil analysis. Part 2: Chemical properties, (3rd ed.) ASA, SSSA, CSSA, Madison, WI.

Swartzendruber, D. (1974). Infiltration of constant-flux rainfall into soil as analysed by the approach of Green and Ampt. Soil Science, 117: 272:281.

Thomas, D.S.G.s and N.J. Middleton (1994). Desertification exploding the myth. John Wiley & Sons, Chichester, England.

Thompson, W.H. and P.C., Robert (1995). Evaluation of mapping strategies for variable rate applications. In: Robert, P. et al. (eds.), Proceedings of Site-Specific Management for Agricultural Systems. 2nd International Conference. March 27-30. Minneapolis, MN, 303-323 pp.

Thornes, J.B. (1989). Geomorphology and grass roots models. In: B. Macmillan (ed.) Remodelling Geography. Blackwell, Oxford, 3-21 pp.

Thornes, J.B (1990). The interaction of erosional and vegetation dynamics in land degradation: spatial outcomes. Vegetation and erosion processes and environments. J.Thornes (ed.). John Wiley & Sons Ltd., Chichester, 41-53 pp.

Thornes, J.B. (1983). Potential and actual erosion around archaeological sites in southeast Spain. Rainfall simulation, runoff and soil erosion. CATENA. Supplement 4, Braunschweig: 91-113.

Thornes, J.B. (1985). The ecology of erosion. Geography, 68 (3): 225-235.

Thornes, J.B. (1995). Mediterranean desertification and the vegetation cover. Desertification in an European context: Physical and socio-economic aspects. R. Fantechi, D. Peter, P. Balabanis and J.L. Rubio (Eds.). European Commission, Brussels, 169-194.

Tisdall, J.M. and J.M. Oades (1982). Organic matter and water-stable aggregates in soils. Journal of Soil Science, 33: 141-163.

Torri, D. and L. Broselli (2000). Water Erosion. In: M.E. Sumner (ed.), Handbook of Soil Science. CRC Press, Boca Raton, Florida. G-171- G-186 pp.

Trimble, S.W. (1990). Geomorphic effects of vegetation cover and management: Some time and space considerations in prediction o erosion and sediment yield. In: Thornes, J. (Ed.), Vegetation and Erosion, Processes and Environments. Wiley, London, 55-65 pp.

Tucker, G.E. and R.L. Bras (1999). Dynamics of Vegetation and Runoff Erosion. Part I-E of final technical report submitted to U.S. Army Corps of Engineers, Construction Engineering Research Laboratory (USACERL). Department of Civil and Environmental Engineering, Massachusetts Institute of Technology, Cambridge, USA. 1-15 pp.

Tunstall, D. (1992). Developing Environmental Indicators: Definitions, framework and issues. (Draft paper). Background Materials for the World Resources Institute. Workshop on Global Environmental Indicators, Washington, D.C., December 7-8, 1992. World Resources Institute, Washington, D.C.

Tunstall, D. (1994). Developing and Using Indicators of Sustainable Development in Africa: An Overview. (Draft paper). Prepared for the Network for Environment and

Sustainable Development in Africa (NESDA). Thematic Workshop on Indicators of Sustainable Development, Banjul, The Gambia, May 16-18, 1994.

Turc, L. (1961). Evaluation des besoins en eau d'irrigation, évapotranspiration potentielle, formulation simplifié et mise à jour. Ann. Agron., 12: 13-49.

UN (1977). Desertification: its causes and consequences (Compiled and edited by the Secretariat of the UN Conference on Desertification, Nairobi). Oxford: Pergamon Press.

UN Center on Transnational Corporations (1985). Environmental aspects of the activities of transnational corporations: a survey. New York: UNCTC.

UN- Secretariat of the Conference on Desertification (1977). Desertification: An overview, In: Desertification: Its causes and consequences. New York, Pergamon Press.

UNCOD (1978). United Nations Conference on Desertification. 29 Aug-9 Sept 1977 Round-up, plan of action and resolutions. UNCOD-UNEP, New York, EUA, 43 p.

UNEP/ FAO (1984). Map of desertification hazards. Explanatory note. UNEP, May 1984.

UNEP/ FAO/ UNESCO/ WMO (1974). Desertification Map of the World, A Conf. 72-2

UNEP/ UN (1990). Global assessment of land degradation/ desertification- GAP II, Desertification Control Bulletin 18, 24-25

USGS (2003). United States Geological Survey. Lake Tahoe Data Clearinghouse: Digital Elevation Model (DEM). http://tahoe.usgs.gov/DEM.html, accessed on May 2003.

Vacca, A., Loddo, S., Ollesch, G., Puddu, R., Serra, G., Tomasi, D., Aru, A. (2000). Measurement of runoff and soil erosion in three areas under different land use in Sardinia (Italy). Catena, 40, 69-92.

Valentin, C. (1994). Surface sealing as affected by various rock fragment covers in West Africa. In: J. Poesen and H. Lavee (Eds.), Rock Fragments in Soil: Surface Dynamics. Catena, 23: 87-97.

Van Deursen, W.P.A. and Wesseling, C.G. (1997). PCRaster Software. Department of Physical Geography, University of Utrecht. http://www.geog.uu.nl/pcraster/pcraster.html, accessed on January 2003.

Vandaele, K. (1993). Assessment of factors affecting ephemeral gully erosion in cultivated catchments of the Belgian Loam Belt. In: Wicherek, S. (ed.), Farm land erosion, Elsevier, Amsterdam, 125-239 pp.

Wainwright, J. (1994). Anthropogenic factors in the degradation of semi-arid lands: a prehistoric case study in Southern France. In: Millington, A.C. and K. Pye (Eds.), Environmental changes in dry lands: biogeochemical and geomorphological perspectives. John Wiley & Sons, London. 427-441 pp.

Walkley-Black, I.A. (1934). An examination of the Degtjaredd method for determining soil organic matter and proposed modification of the chromic titration method. Soil Science, 37: 29-38 pp.

Waller-Hunter, J.H. (1996). Testing Indicators of Sustainable Development: Building on a Strong Foundation. Second Ghent Workshop on Indicators of Sustainable Development.

Ward, M.P.; and T.E. Carpenter (1996). Simulation Modelling of the Effect of Climatic Factors on Bluetongue Virus Infection in Australian Cattle Herds – 1. Model Formulation, Verification and Validation. Preventive Veterinary Medicine, 27(1-2): 1-12.

Warren, A. and C. Agnew (1988). And assessment of desertification and land degradation in arid and semiarid areas. International Institute for Environment and Development paper No. 2. London: Ecology and Conservation Unit, University College.

Warren, A. and J. Maizels (1977). Ecological change and desertification. In: Desertification: its causes and consequences, (ed.), UN. Oxford: Pergamon Press, 160-260 pp.

Wehmeier, E. (1980). Desertification processes and groundwater utilization in the northern Nefzaoua, Tunisia, Stuttgarter Geographische Studien, 95, 125-43 pp.

Winograd, M. (1997). Vertical and horizontal linkages in the context of indicators of sustainable development. In: Sustainability indicators: Report of the project on Indicators of Sustainable Development (SCOPE). Moldran, B.; Billharz, S,: Matravers, R. (eds.). John Wiley & Sons, Chichester, UK. 92-95 pp.

Winograd, M. and Eade, J. (1997). Environmental and sustainability indicators for Latin America and the Caribbean: the use of geographical information systems (GIS). In: Moldan, B.; Billharz, S.; Matravers, R. (eds.), Sustainability Indicators: Report of the project on Indicators of Sustainable Development (SCOPE). John Wiley & Sons.

Wischmeier, W.H., and D.D. Smith (1978). Predicting Rainfall Erosion Losses: a Guide to Conservation Planning. U.S. Department of Agriculture, Washington, D.C., 58 p.

Wischmeier, W.H., and J.V. Mannering (1969). Relation of soil properties to its erodivility. Soil Sci. Soc. Am. Proc. 33: 131-137.

Wischmeier, W.H., Johnson, C.B., Cross, B.V. (1971). A soil erodability monograph for farmland and construction sites. Journal of Soil and Water Conservation, 26: 189-193.

Wolman, M.G. and F.G. Fournier (eds.) (1987). Land transformation in Agriculture. Chichester: John Wiley & Sons.

World Bank, The (1995). Monitoring Environmental Progress. The World Bank, Washington, D.C.

World Meteorological Organization (WMO) (1980). Expert Meeting on Meteorological aspects of Desertification, December 1980. Report of the Meeting, point 2.4. Geneva, WMO.

World Meteorological Organization (WMO) (1983). Meteorological aspects of certain processes affecting soil degradation – especially erosion. Tech. Note no 17, WMO n. 159:149

Wotawa, G., Stohl, A., KrompKolb, H. (1997). Estimating the Uncertainty of a Lagrangian Photochemical Air Quality Simulation Model caused by Inexact Meteorological Input Data. Reliability Engineering & System Safety, 57(1): 31-40.

Yair, A. and H. Lavee (1976). Runoff generative process and runoff yield from arid talus mantles slopes. Earth Surface Processes, 1: 235-247.

Zanchi, C. (1993). Aspetti dell'erosione dei suoli nei diversi ambienti del bacino del Mediterraneo, in montagna ed in collina. In: Atti del Convegno La difesa del suolo in ambiente mediterraneo, pp. 57-69. Cala Gonone 12-14 giugno 1991, ERSAT, Cagliari.

Zhang, X. (1999). Soil-erosion modelling at the global scale using remote sensing and GIS. Unpublished Thesis. University of London.

**PROCEDURE OF THE CALCULATION OF OVERLAND FLOW IN PCRASTER**

First of all the calculation of the local drainage directions map (LDD) of the study catchment was created on the basis of the Digital Elevation Model (DEM) through the *llddcreate* PCRaster operator. This function creates a local drain direction map using the 8 point pour algorithm with flow directions from each cell to its steepest downslope neighbour. The operator determines for each cell its neighbouring cell to which runoff will flow to. It first calculates the LDD of each cell of the map and then links them obtaining a local drain direction network, the flow pattern on the map.

$LDD_1$ = lddcreate (DEM, 1e31, 1e31, 1e31, 1e31)

$LDD_1$ = local drain direction map of the study area
DEM = digital elevation model of the study area

Each cell on a Ldd must have pit at the end of its downstream path, and if this is not the case for one or more cells on a Ldd, the map is considered unsound. Therefore, the l*ddrepair* operator, was used for repairing the Ldd by changing the cell values on the unsound LDD in such a way that it becomes sound, with all downstream paths ending in a pit cell.

$LDD_2$ = lddrepair ($LDD_1$)

$LDD_2$ = repaired local drain direction map of the study area
$LDD_1$ = local drain direction map of the study area

The last step in the creation of the LDD is the interactive removal of the pits. A pit is a cell whose neighbours all have a local drainage direction in direction of the pit cell. A pit cell does not have a local drain direction because all its neighbours are at a higher elevation. Additionally the outflow cell of each catchment on the map, which is a cell at the edge of the map is also a pit. The *pit* operator is used for identifying where all the pits are in the created LDD.

Pits = pit ($LDD_3$)

Pits = map of the pits found in the local drain direction map
$LDD_3$ = local drain direction map of the study area, repaired and without pits

The production of the drainage direction map is preprocessing whilst the accuthresholdflux is applied for every timestep.

Then, once the local drainage direction map was properly created, the overland flow was calculated, through the accuthresholdflux operator, and using the LDD, and the outcomes of the runoff and actual infiltration capacity sub-model:

$OFLOW_{(t)}$ = accuthresholdflux (LDD, $Runoff_{(t)}$, $AINF_{(t)}$)

OFLOW = overland flow computed from the runoff routing (mm water/ month)
LDD = local drain direction map
AINF = soil actual infiltration capacity (mm water/ month)
Runoff = runoff water generation (mm water/ month)

This overland flow function can be described by flow of runoff through a set of linked systems, where a cell represents a system. The flow starts at the cells/systems at the watershed boundaries (defined by LDD) and ends at a pit cell. The systems are linked by the local drain directions, these define the path of flow through the set of cells/ systems. Each time a system is passed, the amount of flow changes.

**PROCEDURE FOR CALCULATING SOLAR RADIATION IN PCRASTER, AS A PART OF THE POTENTIAL EVAPOTRANSPIRATION SUB-MODEL**

The first step for estimating the potential evapotranspiration (PET) through the Priestley-Taylor method was to calculate the solar radiation at the land surface (RADSPT). This procedure consisted of several steps:

1. Simulation of monthly solar radiation maps at the top of the atmosphere through the previously developed digital elevation model (DEM) of the study catchment. In total 12 maps were created, one per each month of the year.

2. Through the Solargen PCRaster model (Mulligan, 1996), compute the solar radiation at the top of the atmosphere on the basis of the latitude and longitude of the study area.

3. Calculate for each month of the year, the difference in percentage between the solar radiation at the top of the atmosphere from the Solargen model, and the solar radiation at the land surface obtained from the Automatic Weather Station (AWS). The value measured at the weather station should be around 50% lower because of losses in atmospheric transmission losses.

4. Apply each of the previously calculated difference percentages, to the simulated maps of solar radiation at the top of the atmosphere, in order to create the correspondent maps of solar radiation at the land surface. Note this takes no account of any spatial (altitudinal) variation in cloud cover.

5. Once the monthly solar radiation maps at the land surface (RADS) have been created, through the *timeinput operator*, the correspondent map at every timestep in the model is made available for the programme and is renamed as RADSPT:

RADSPT = timeinput (RADS)

RADSPT = monthly daily average solar radiation map at the land surface ($MJ/m^2$)
RADS = monthly daily average solar radiation map at the land surface ($MJ/m^2$)

6. Afterwards, the net solar radiation (RADNET) is computed on the basis of the RADSPT monthly maps, and the correspondent albedo value per each specific selected land use.

RADNET = (RADSPT * (1.0 - albedo))

RADNET = net solar radiation monthly maps ($MJ/m^2$)
RADSPT = monthly daily average solar radiation map at the land surface ($MJ/m^2$)
albedo = ratio of outgoing solar radiation from the snow or ground to the incoming solar radiation from the atmosphere. Specific values per each land use were employed based on literature review values (dimensionless). Here, it has to be taken into consideration that this is a simplification because net radiation is not just solar in minus solar reflected but solar plus longwave in minus solar reflected plus longwave emitted.

## PROCEDURE FOR THE CALCULATION OF SATURATION VAPOUR PRESSURE CURVE IN PCRASTER, AS A PART OF THE POTENTIAL EVAPOTRANSPIRATION SUB-MODEL

The slope of saturation vapour pressure curve was another parameter required in the Priestley-Taylor equation, and it was calculated as follows:

1. Use of the *timeinputscalar operator*, which allows reading at every timestep of the model a particular value from a column in a text file, and in this case the data is the monthly temperature:

TempMonth = timeinputscalar (TempTimeSeries, 1)

TempMonth = mean temperature related to the month of the timestep in the model (ºC)
TempTimeSeries = time series file the monthly temperature data in a column, sequentially ordered starting in January.

Here, the "1" in the equation refers to the values of interest placed in the column number one in the text file.

2. Once the monthly temperature data is available, the slope of saturation vapour is directly calculated as a function of the temperature, through the following equation:

$$\text{Delta} = (33.8639 * 0.05904 * ((0.00738 * \text{TempMonth}) + 0.8072)^7 - 0.0000342))$$

Delta = slope of saturation vapour pressure curve (KPa/ºC)
TempMonth = mean temperature related to the month of the timestep in the model (ºC)

# SCRIPT OF THE DESERTIFICATION INDICATOR SYSTEM (DIS) IN PCRASTER

```
## Rainfall scenario: Dry year

binding

#------------------------Soil Moisture Model---------------------------------------------

Moisture = MoistSim120.map;     # Soil moisture content per land use.
                                # in mmwater/mmsoil.

Porosity = POROS2.map;          # Soil porosity map of each land use.
                                # in mmporespace/mmsoil.

SoilDepth = SDepth2bis.map;     # Initial soil depth map per each land use (Roads = 0.001).
                                # in mm soil.
#--------------------------------

Moist = Moist;                  # Soil Moisture output maps computed per each timestep.
                                # in m3water/m3soil.

#Depth = Depth;                 # Soil depth output maps computed per each timestep.
                                # in mm soil.
#--------------------------------

AIN = scalar (0);               # Initial Actual Infiltration value which is going to be declared in the
                                # initial section as a input in the first timestep in the dynamic section.
                                # in mm/month.

AET = scalar (0);               # Initial Actual Evapotranspiration value which is going to be declared in
```

```
                                # the initial section as a input in the first timestep in the dynamic section.
                                # in mm/month.

RECH = scalar (0);              # Initial Recharge value which is going to be declared in the initial section
                                # as a input in the first timestep in the dynamic section.
                                # in mm/month.
#--------------------------------

RFMoist = RFMoist;              # Water surplus (when soil moisture exceeds the soil porosity).
                                # in mm/month.

#-----------------------Potential Infiltration Model-------------------------------------

#Moist = Moist;

Pot = Pot;

PotInfil = PotInfil;            # Field potential infiltration capacity per land use
                                # in mm/hr.

h = 25;

SatHK = SatHK2.map;             #Saturated hydraulic conductivity.
                                # in mm/hr.


#-----------------------Precipitation----------------------------------------------------

RainTimeSeries = DryRain120.tss;    # Time Series file (tss) of monthly rainfall amount.
                                    # in mm/month.
```

```
Rain = Rain;                          # Rainfall values from the Time Series file per each timestep (= month).
                                      # in mm/month.

#------------------------Runoff Model-----------------------------------------------------

Runoff = Runoff;                      # Runoff + water surplus from the soil moisture model.
                                      # in mmwater/month.

#SumRunoff = SumRunoff;                # Runoff output maps per each timestep
                                      # in mm/month.

#------------------------Actual Infiltration Model-----------------------------------------

AINF = AINF;                          # Actual infiltration output maps per each timestep.
                                      #in mm/month.

#---------------------Runoff Routing: accuthresholdflux operator------------------------

OFLOW = OFLOW;                        # Monthly overland flow (Runoff) based on the local drainage direction map,
                                      # in mm/month.

DEM = Dem.map;                        # Digital Elevation Model map of the study area.
                                      # in meters.

LDD = LDD.map;                        # Local drain direction map, computed from the DEM map.


Pits = Pits.map;

#Runoff = Runoff;
```

```
#AINF = AINF;

#-----Potential Evapotranspiration Model: Priestley - Taylor Method (1972)----------

TempTimeSeries = DryTemp120.tss;     # Time series file of monthly temperature.
                                     # in ºC/month.

Landuse = Landuse.map;               # Land use map of the study catchment (10 different land uses)

RADS = RADS;                         # Simulated INPUT  Solar radiation maps at the land surface
                                     # in MJ/m2/month (Daily mean values of each month).

RADSPT = RADSPT;                     # Result maps per each timestep using the "timeinput" operator on the RADS maps
                                     # in MJ/m2/month.

TempMonth = TempMonth;               # ºC/month.

RADNET = RADNET;                     # Output Net solar radiation maps at the land surface.
                                     # in MJ/m2/month.

Lambda = 2.45;                       #Latent heat of vaporization. This value corresponds to a temperature of 20ºC
                                     #in MJ/Kg.

PTcoeff = 1.28;                      # Priestley-Taylor constant. Dimensionless.

Psychro = 1.26;                      # Constant, in KPa/ºC.

Delta = Delta;                       # Slope of saturation vapour pressure curve, in KPa/ºC.

PET = PET;                           # Potential Evapotranspiration output maps per each timestep.
                                     # in mm/month.
```

```
#------Actual Evapotranspiration Model---------------------------------------------------------------

ActET = ActET;                  # Actual Evapotranspiraion output maps per each timestep
                                # in mm/month.

#PET = PET;

#Moist = Moist;

#Porosity = POROS2.map;

#Depth = Depth;

#---------------Recharge Model: from the "Solargen" Spreadsheet Model-----------------------

#SatHK = SatHK2.map;            # Map of the saturated hydraulic conductivity per each land use (Roads = 0)
                                # in mm/hr.

SatMoist = scalar (1);          # Saturated volumetric water content, constant.
                                # in m3water/m3soil.

b = bvalues2.map;               # b values per each land use (Roads = 0).
                                # dimensionless.

#GRECH = GRECH;                 # Groundwater recharge
                                # in mm/hr.

Recharge = Recharge;            # Groundwater recharge in mm/month.
                                # in mm/hr.

#Moist = Moist;
```

```
#-------Soil Erosion Model: based on John Thornes's Wash Erosion Model (1990)---------------

EROS = EROS;                  #Soil erosion output maps, directly computed from the Thorne's Model.
                              # in mm/month.

Erosion = Erosion;            #Soil erosion output maps including the restriction that the eroded soil
                              #cannot be greater than the soil depth.
                              # in mm/month.

Depth = Depth;                # Output maps of the soil depth per each timestep.
                              # in mm soil.

K = scalar (2.0);             # Soil erodability factor.

#OFLOW = OFLOW;

M = scalar (1.66);            # Discharge scaling factor.

Slopedeg = Slopedeg.map;      # Slope map of the study area computed from the DEM map.
                              # in degrees.

# tan(Slopedeg): this calculates the tangent of the slope map.

N = scalar (2.0);             # Slope constant.

Vegetation = VEG2.map;        # Map of the % of vegetation cover of each land use in the study area.

#------------------------Clone Map = Reference map-----------------------------------------

Clone = Cloneboolean.map;     # Boolean clone map of the overall catchment area.
```

```
Clonescalar = Clonescalar.map;          # Scalar clone map of the overall catchment area.

#####################################################################################################
#VERIFICATION OF THE MODEL####################################################################
#####################################################################################################

Lusenom = lusenom.map;                  #Landuse nominal map from the scalar Landuse.map

ActiveCells = ActiveCells.map;          #Total number of cells composing the study area.

LuseArea = LuseArea.map;                #Total number of cells composing each land use in the study area.

#DIRECT OUTPUTS OF THE MODEL#################################################################
#####################################################################################################

#MOISTURE SUBMODEL'S OUTPUTS################################################################

MoistTSS = MoistTSS.tss;                #Maptotal operator: Soil Moisture content results are the
                                        #average of the values per cell and timestep.


MoistureTSS = MoistureTSS.tss;          #Timeoutput operator: Soil Moisture content results are the
                                        # average of values related to each land use on a timestep basis.

#POTENTIAL INFILTRATION SUBMODEL'S OUTPUTS#################################################

PInfilTSS = PInfilTSS.tss;              #Maptotal operator: Potential Infiltration results are the
                                        #average of the values per cell and timestep.

PotInfilTSS = PotInfilTSS.tss;          #Timeoutput operator: Potential Infiltration results are the average of
                                        # values related to each land use on a timestep basis.
```

```
#RUNOFF SUBMODEL'S OUTPUTS#########################################

RoffTSS = RoffTSS.tss;            #Maptotal operator: Runoff results are the
                                  #average of the values per cell and timestep.

RunoffTSS = RunoffTSS.tss;        #Timeoutput operator: Runoff results are the average of
                                  # values related to each land use on a timestep basis.

#ACTUAL INFILTRATION SUBMODEL'S OUTPUTS###############################

AInfTSS = AInfTSS.tss;            #Maptotal operator: Actual Infiltration results are the
                                  #average of the values per cell and timestep.

AInfilTSS = AInfilTSS.tss;        #Timeoutput operator: Actual Infiltration results are the average of
                                  # values related to each land use on a timestep basis.

#RUNOFF ROUTING SUBMODEL'S OUTPUTS##################################

OflowTSS = OflowTSS.tss;          #Timeoutput operator: Overland flow results are the average of
                                  # values related to each land use on a timestep basis.

#POTENTIAL EVAPOTRANSPIRATION SUBMODEL'S OUTPUTS######################

PETTSS = PETTSS.tss;              #Maptotal operator: Potential Evapotranspiration results are the
                                  #average of the values per cell and timestep.

PotEvapTSS = PotEvapTSS.tss;      #Timeoutput operator: Potential Evapotranspiration results are the average of
                                  # values related to each land use on a timestep basis.
```

```
#ACTUAL EVAPOTRANSPIRATION SUBMODEL'S OUTPUTS#######################################

AETTSS = AETTSS.tss;                 #Maptotal operator: Actual Evapotranspiration results are the
                                     #average of the values per cell and timestep.

ActEvapTSS = ActEvapTSS.tss;         #Timeoutput operator: Actual Evapotranspiration results are the average of
                                     # values related to each land use on a timestep basis.

#RECHARGE SUBMODEL'S OUTPUTS###################################################################

RechTSS = RechTSS.tss;               #Maptotal operator: Recharge results are the
                                     #average of the values per cell and timestep.

RechargeTSS = RechargeTSS.tss;       #Timeoutput operator: Recharge results are the average of
                                     # values related to each land use on a timestep basis.

#SOIL EROSION SUBMODEL'S OUTPUTS################################################################

ErosionTSS = ErosionTSS.tss;         Maptotal operator: Soil Erosion results are the
                                     #average of the values per cell and timestep.

SoilErosionTSS = SoilErosionTSS.tss; #Timeoutput operator: Soil Erosion results are the average of
                                     # values related to each land use on a timestep basis.

#INDIRECT OUTPUTS OF THE MODEL##################################################################
###########################################################################################################

#RFMoistTSS = RFMoistTSS.tss;

#RoffMoistTSS = RoffMoistTSS.tss;
```

```
#SumRoffTSS = SumRoff.tss;
#SumRunoffTSS = SumRunoffTSS.tss;
DepthTSS = DepthTSS.tss;
SoilDepthTSS = SoilDepthTSS.tss;
#GRECHTSS = GRECHTSS.tss;
#GRechargeTSS = GRechargeTSS.tss;
ErosTSS = ErosTSS.tss;
SErosTSS = SErosTSS.tss;
##########################################################################################
areamap
Clone;
timer
1 24 1;                        # 24 timesteps = 2 years * 12 month/year.
initial
#------------------------Soil Moisture Model-----------------------------------------------
Moist = Moisture;              # Moisture values for the first timestep in the dynamic section which are going to
```

```
                                   # be taken from the "Moisture.map" declared as "Moisture" in the binding section.
                                   # in m3water/m3soil.

Depth = SoilDepth;                 # Soil depth values for the first timestep in the dynamic section are going to be
                                   # those from the "SDepth.map" declared as "SoilDepth" in the binding section.
                                   # in mm soil.

AINF = AIN;                        # Initial actual infiltration value which is going to be used as input in the first
                                   # timestep in the dynamic section of the Soil Moisture Model.
                                   # in mm/month.

ActET = AET;                       # Initial actual evapotranspiration value which is going to be use as input in the
                                   # first timestep in the dynamic section of the Soil Moisture Model.
                                   # in mm/month.

Recharge = RECH;                   # Initial recharge value which is going to be use as input in the first timestep in
                                   # the dynamic section of the Soil Moisture Model.
                                   # in mm/month.

#----------------Runoff Routing: accuthresholdflux operator-------------------------

 Ldd = lddcreate (DEM, 1e31, 1e31, 1e31, 1e31);

report LDD = lddrepair (Ldd);

 report Pits = pit (LDD);

##############################################################################################
#######################VERIFICATION OF THE MODEL:#########################################

#report ActiveCells = maptotal (Clonescalar);     #Calcul of the total number of cells composing the overall study area.
```

```
ActiveCells = maptotal (Clonescalar);


#report LuseArea = areatotal (Clonescalar, Lusenom);        #Calcul of the total number of cells which compose each land use
                                                             #The sum of all the values of each land use equals to the total number
                                                             #of cells of the overall study area.

LuseArea = areatotal (Clonescalar, Lusenom);

##########################################################################################

dynamic

#------------------------Soil Moisture Model----------------------------------------------

Moist = if (Landuse eq 9 then 0 else (Moist + ((AINF - ActET - Recharge) div (Depth))));        # in mmwater/mmsoil

#-------------------------------

Moist = if ((Moist gt Porosity) then Porosity else Moist);        #in mmwater/mmsoil

#-------------------------------

report Moist = if (Moist le 0.001 then 0.001 else Moist);        #in mmwater/mmsoil
#-------------------------------

RFMoist = if ((Moist gt Porosity) then ((Moist - Porosity) * Depth) else 0);        #in mmwater/month

##########################################################################################

#------------------------Potential Infiltration Model--------------------------------------
```

```
Inf1 = if (Landuse eq 1 then 1132.524 else 0);
Inf2 = if (Landuse eq 2 then 2900.649 else 0);
Inf3 = if (Landuse eq 3 then 2571.080 else 0);
Inf4 = if (Landuse eq 4 then 1776.197 else 0);

Inf5 = if (Landuse eq 5 then 401.217 else 0);
Inf6 = if (Landuse eq 6 then 773.592 else 0);
Inf7 = if (Landuse eq 7 then 588.128 else 0);
Inf8 = if (Landuse eq 8 then 3447.103 else 0);
Inf9 = if (Landuse eq 9 then 0 else 0);
Inf10 = if (Landuse eq 10 then 431.969 else 0);

Pot = Inf1+Inf2+Inf3+Inf4+Inf5+Inf6+Inf7+Inf8+Inf9+Inf10; #Field work Average Infiltration in mmwater/hour.

report PotInfil = SatHK + (Pot - SatHK) * exp (- h * Moist);          #mmwate/hour. Horton's equation.

##############################################################################################

#------------------------Precipitation----------------------------------------------------

Rain = timeinputscalar (RainTimeSeries, 1);          # in mm/month.

 Rain1mmhr = (Rain * 0.1450);
 Rain2mmhr = (Rain * 0.0812);
 Rain3mmhr = (Rain * 0.0978);
 Rain4mmhr = (Rain * 0.0413);
 Rain5mmhr = (Rain * 0.0213);
 Rain6mmhr = (Rain * 0.1218);
 Rain8mmhr = (Rain * 0.0206);
 Rain12mmhr = (Rain * 0.1969);
 Rain18mmhr = (Rain * 0.0040);
```

```
Rain24mmhr = (Rain * 0.1464);
Rain36mmhr = (Rain * 0.0645);
Rain48mmhr = (Rain * 0.0120);
Rain60mmhr = (Rain * 0.0140);
Rain72mmhr = (Rain * 0.0140);
Rain84mmhr = (Rain * 0.0027);
Rain96mmhr = (Rain * 0.0040);
Rain108mmhr = (Rain * 0.0033);
Rain132mmhr = (Rain * 0.0007);
Rain144mmhr = (Rain * 0.0027);
Rain156mmhr = (Rain * 0.0013);
Rain180mmhr = (Rain * 0.0007);
Rain192mmhr = (Rain * 0.0020);
Rain204mmhr = (Rain * 0.0007);
Rain264mmhr = (Rain * 0.0007);
Rain276mmhr = (Rain * 0.0007);

##############################################################################################

#-----------------------Runoff Model------------------------------------------------------

RmI1mmhr= (1 - PotInfil);
RmI2mmhr= (2 - PotInfil);
RmI3mmhr= (3 - PotInfil);
RmI4mmhr= (4 - PotInfil);
RmI5mmhr= (5 - PotInfil);
RmI6mmhr= (6 - PotInfil);
RmI8mmhr= (8 - PotInfil);
RmI12mmhr= (12 - PotInfil);
RmI18mmhr= (18 - PotInfil);
RmI24mmhr= (24 - PotInfil);
```

```
RmI36mmhr= (36 - PotInfil);
RmI48mmhr= (48 - PotInfil);
RmI60mmhr= (60 - PotInfil);
RmI72mmhr= (72 - PotInfil);
RmI84mmhr= (84 - PotInfil);
RmI96mmhr= (96 - PotInfil);
RmI108mmhr= (108 - PotInfil);
RmI132mmhr= (132 - PotInfil);
RmI144mmhr= (144 - PotInfil);
RmI156mmhr= (156 - PotInfil);
RmI180mmhr= (180 - PotInfil);
RmI192mmhr= (192 - PotInfil);
RmI204mmhr= (204 - PotInfil);
RmI264mmhr= (264 - PotInfil);
RmI276mmhr= (276 - PotInfil);

R1mmhr = if (RmI1mmhr gt 0 then RmI1mmhr else 0);
R2mmhr = if (RmI2mmhr gt 0 then RmI2mmhr else 0);
R3mmhr = if (RmI3mmhr gt 0 then RmI3mmhr else 0);
R4mmhr = if (RmI4mmhr gt 0 then RmI4mmhr else 0);
R5mmhr = if (RmI5mmhr gt 0 then RmI5mmhr else 0);
R6mmhr = if (RmI6mmhr gt 0 then RmI6mmhr else 0);
R8mmhr = if (RmI8mmhr gt 0 then RmI8mmhr else 0);
R12mmhr = if (RmI12mmhr gt 0 then RmI12mmhr else 0);
R18mmhr = if (RmI18mmhr gt 0 then RmI18mmhr else 0);
R24mmhr = if (RmI24mmhr gt 0 then RmI24mmhr else 0);
R36mmhr = if (RmI36mmhr gt 0 then RmI36mmhr else 0);
R48mmhr = if (RmI48mmhr gt 0 then RmI48mmhr else 0);
R60mmhr = if (RmI60mmhr gt 0 then RmI60mmhr else 0);
R72mmhr = if (RmI72mmhr gt 0 then RmI72mmhr else 0);
R84mmhr = if (RmI84mmhr gt 0 then RmI84mmhr else 0);
```

```
R96mmhr = if (RmI96mmhr gt 0 then RmI96mmhr else 0);
R108mmhr = if (RmI108mmhr gt 0 then RmI108mmhr else 0);
R132mmhr = if (RmI132mmhr gt 0 then RmI132mmhr else 0);
R144mmhr = if (RmI144mmhr gt 0 then RmI144mmhr else 0);
R156mmhr = if (RmI156mmhr gt 0 then RmI156mmhr else 0);
R180mmhr = if (RmI180mmhr gt 0 then RmI180mmhr else 0);
R192mmhr = if (RmI192mmhr gt 0 then RmI192mmhr else 0);
R204mmhr = if (RmI204mmhr gt 0 then RmI204mmhr else 0);
R264mmhr = if (RmI264mmhr gt 0 then RmI264mmhr else 0);
R276mmhr = if (RmI276mmhr gt 0 then RmI276mmhr else 0);

Run1mm = if (R1mmhr eq 0 then 0 else Rain1mmhr);
Run2mm = if (R2mmhr eq 0 then 0 else Rain2mmhr);
Run3mm = if (R3mmhr eq 0 then 0 else Rain3mmhr);
Run4mm = if (R4mmhr eq 0 then 0 else Rain4mmhr);
Run5mm = if (R5mmhr eq 0 then 0 else Rain5mmhr);
Run6mm = if (R6mmhr eq 0 then 0 else Rain6mmhr);
Run8mm = if (R8mmhr eq 0 then 0 else Rain8mmhr);
Run12mm = if (R12mmhr eq 0 then 0 else Rain12mmhr);
Run18mm = if (R18mmhr eq 0 then 0 else Rain18mmhr);
Run24mm = if (R24mmhr eq 0 then 0 else Rain24mmhr);
Run36mm = if (R36mmhr eq 0 then 0 else Rain36mmhr);
Run48mm = if (R48mmhr eq 0 then 0 else Rain48mmhr);
Run60mm = if (R60mmhr eq 0 then 0 else Rain60mmhr);
Run72mm = if (R72mmhr eq 0 then 0 else Rain72mmhr);
Run84mm = if (R84mmhr eq 0 then 0 else Rain84mmhr);
Run96mm = if (R96mmhr eq 0 then 0 else Rain96mmhr);
Run108mm = if (R108mmhr eq 0 then 0 else Rain108mmhr);
Run132mm = if (R132mmhr eq 0 then 0 else Rain132mmhr);
Run144mm = if (R144mmhr eq 0 then 0 else Rain144mmhr);
Run156mm = if (R156mmhr eq 0 then 0 else Rain156mmhr);
```

```
Run180mm = if (R180mmhr eq 0 then 0 else Rain180mmhr);
Run192mm = if (R192mmhr eq 0 then 0 else Rain192mmhr);
Run204mm = if (R204mmhr eq 0 then 0 else Rain204mmhr);
Run264mm = if (R264mmhr eq 0 then 0 else Rain264mmhr);
Run276mm = if (R276mmhr eq 0 then 0 else Rain276mmhr);

SumRunoff = Run1mm+Run2mm+Run3mm+Run4mm+Run5mm+Run6mm+Run8mm+Run12mm+
Run18mm+Run24mm+Run36mm+Run48mm+Run60mm+Run72mm+Run84mm+Run96mm+
Run108mm+Run132mm+Run144mm+Run156mm+Run180mm+Run192mm+Run204mm+Run264mm+Run276mm;

report Runoff = SumRunoff + RFMoist;     # in mmwater/month

############################################################################################

#------------------------Actual Infiltration Model---------------------------

AINF1mm = if (RmI1mmhr le 0 then Rain1mmhr else 0);
AINF2mm = if (RmI2mmhr le 0 then Rain2mmhr else 0);
AINF3mm = if (RmI3mmhr le 0 then Rain3mmhr else 0);
AINF4mm = if (RmI4mmhr le 0 then Rain4mmhr else 0);
AINF5mm = if (RmI5mmhr le 0 then Rain5mmhr else 0);
AINF6mm = if (RmI6mmhr le 0 then Rain6mmhr else 0);
AINF8mm = if (RmI8mmhr le 0 then Rain8mmhr else 0);
AINF12mm = if (RmI12mmhr le 0 then Rain12mmhr else 0);
AINF18mm = if (RmI18mmhr le 0 then Rain18mmhr else 0);
AINF24mm = if (RmI24mmhr le 0 then Rain24mmhr else 0);
AINF36mm = if (RmI36mmhr le 0 then Rain36mmhr else 0);
AINF48mm = if (RmI48mmhr le 0 then Rain48mmhr else 0);
AINF60mm = if (RmI60mmhr le 0 then Rain60mmhr else 0);
AINF72mm = if (RmI72mmhr le 0 then Rain72mmhr else 0);
```

```
AINF84mm = if (RmI84mmhr le 0 then Rain84mmhr else 0);
AINF96mm = if (RmI96mmhr le 0 then Rain96mmhr else 0);
AINF108mm = if (RmI108mmhr le 0 then Rain108mmhr else 0);
AINF132mm = if (RmI132mmhr le 0 then Rain132mmhr else 0);
AINF144mm = if (RmI144mmhr le 0 then Rain144mmhr else 0);
AINF156mm = if (RmI156mmhr le 0 then Rain156mmhr else 0);
AINF180mm = if (RmI180mmhr le 0 then Rain180mmhr else 0);
AINF192mm = if (RmI192mmhr le 0 then Rain192mmhr else 0);
AINF204mm = if (RmI204mmhr le 0 then Rain204mmhr else 0);
AINF264mm = if (RmI264mmhr le 0 then Rain264mmhr else 0);
AINF276mm = if (RmI276mmhr le 0 then Rain276mmhr else 0);

report AINF = AINF1mm+AINF2mm+AINF3mm+AINF4mm+AINF5mm+AINF6mm+AINF8mm+AINF12mm+
AINF18mm+AINF24mm+AINF36mm+AINF48mm+AINF60mm+AINF72mm+AINF84mm+AINF96mm+AINF108mm+
AINF132mm+AINF144mm+AINF156mm+AINF180mm+AINF192mm+AINF204mm+AINF264mm+AINF276mm;
# in mmwater/month.

#################################################################################################

#-------Overland flow: Runoff routing with the accuthresholdflux operator-------------

report OFLOW = accuthresholdflux (LDD, Runoff, AINF);          # mm/month

#################################################################################################

#--------Potential Evapotranspiration Model: Priestley - Taylor Method (1972)--------

RADSPT= timeinput (RADS);                                      # in MJ/m2 month (daily mean values of each month).

TempMonth = timeinputscalar (TempTimeSeries, 1);               # in ºC/month
```

```
RADNET = if (Landuse eq 1 then (RADSPT * (1.0 - 0.16)) else 0);
RADNET = if (Landuse eq 2 then (RADSPT * (1.0 - 0.17)) else RADNET);
RADNET = if (Landuse eq 3 then (RADSPT * (1.0 - 0.24)) else RADNET);
RADNET = if (Landuse eq 4 then (RADSPT * (1.0 - 0.23)) else RADNET);
RADNET = if (Landuse eq 5 then (RADSPT * (1.0 - 0.17)) else RADNET);
RADNET = if (Landuse eq 6 then (RADSPT * (1.0 - 0.18)) else RADNET);
RADNET = if (Landuse eq 7 then (RADSPT * (1.0 - 0.15)) else RADNET);
RADNET = if (Landuse eq 8 then (RADSPT * (1.0 - 0.16)) else RADNET);
RADNET = if (Landuse eq 9 then (RADSPT * (1.0 - 1)) else RADNET);
RADNET = if (Landuse eq 10 then (RADSPT * (1.0 - 0.15 )) else RADNET);          # in MJ/m2 month

Delta = (33.8639 * 0.05904 * (((0.00738 * (TempMonth)) + 0.8072) ** 7) - 0.0000342);  # in KPa/ºC

report PET = if (Landuse eq 9 then 0 else (PTcoeff * (RADNET div Lambda) * (Delta div (Delta + Psychro)))); # in mmw/month

################################################################################################

#---Actual Evapotranspiration Model: based on the "Solargen" Spreadsheet Model------

report ActET = if (Landuse eq 9 then 0 else (PET * Moist));          # in mm/month.

################################################################################################

#----------Recharge Model: from the "Solargen" Spreadsheet Model-------------------

Moist10 = ((Moist * 10) div 100);
Moist20 = ((Moist * 20) div 100);
Moist30 = ((Moist * 30) div 100);
Moist40 = ((Moist * 40) div 100);
Moist50 = ((Moist * 50) div 100);
Moist60 = ((Moist * 60) div 100);
```

```
Moist70 = ((Moist * 70) div 100);
Moist80 = ((Moist * 80) div 100);
Moist90 = ((Moist * 90) div 100);
Moist100 = ((Moist * 100) div 100);                #mmwater/mmsoil.

GRECH10 = (SatHK * ((Moist10 div SatMoist) ** ((2 * b) + 3)));
GRECH20 = (SatHK * ((Moist20 div SatMoist) ** ((2 * b) + 3)));
GRECH30 = (SatHK * ((Moist30 div SatMoist) ** ((2 * b) + 3)));
GRECH40 = (SatHK * ((Moist40 div SatMoist) ** ((2 * b) + 3)));
GRECH50 = (SatHK * ((Moist50 div SatMoist) ** ((2 * b) + 3)));
GRECH60 = (SatHK * ((Moist60 div SatMoist) ** ((2 * b) + 3)));
GRECH70 = (SatHK * ((Moist70 div SatMoist) ** ((2 * b) + 3)));
GRECH80 = (SatHK * ((Moist80 div SatMoist) ** ((2 * b) + 3)));
GRECH90 = (SatHK * ((Moist90 div SatMoist) ** ((2 * b) + 3)));
GRECH100 = (SatHK * ((Moist100 div SatMoist) ** ((2 * b) + 3)));

Moist10change = (Moist10);
Moist20change = (Moist20 - Moist10);
Moist30change = (Moist30 - Moist20);
Moist40change = (Moist40 - Moist30);
Moist50change = (Moist50 - Moist40);
Moist60change = (Moist60 - Moist50);
Moist70change = (Moist70 - Moist60);
Moist80change = (Moist80 - Moist70);
Moist90change = (Moist90 - Moist80);
Moist100change = (Moist100 - Moist90);             #mmwater/mmsoil

Moist10mm = (Moist10change * Depth);
Moist20mm = (Moist20change * Depth);
Moist30mm = (Moist30change * Depth);
Moist40mm = (Moist40change * Depth);
```

```
Moist50mm = (Moist50change * Depth);
Moist60mm = (Moist60change * Depth);
Moist70mm = (Moist70change * Depth);
Moist80mm = (Moist80change * Depth);
Moist90mm = (Moist90change * Depth);
Moist100mm = (Moist100change * Depth);          # in mmwater

RechTime10 = if (GRECH10 ne 0 then (Moist10mm / GRECH10) else 0);
RechTime20 = if (GRECH20 ne 0 then (Moist20mm / GRECH20) else 0);
RechTime30 = if (GRECH30 ne 0 then (Moist30mm / GRECH30) else 0);
RechTime40 = if (GRECH40 ne 0 then (Moist40mm / GRECH40) else 0);
RechTime50 = if (GRECH50 ne 0 then (Moist50mm / GRECH50) else 0);
RechTime60 = if (GRECH60 ne 0 then (Moist60mm / GRECH60) else 0);
RechTime70 = if (GRECH70 ne 0 then (Moist70mm / GRECH70) else 0);
RechTime80 = if (GRECH80 ne 0 then (Moist80mm / GRECH80) else 0);
RechTime90 = if (GRECH90 ne 0 then (Moist90mm / GRECH90) else 0);
RechTime100 = if (GRECH100 ne 0 then (Moist100mm / GRECH100) else 0);

RechTimeTotal = (RechTime10+RechTime20+RechTime30+RechTime40+RechTime50+RechTime60
+RechTime70+RechTime80+RechTime90+RechTime100);

RechFrac10 = if (RechTime10 ne 0 then (RechTime10 / RechTimeTotal) else 0);
RechFrac20 = if (RechTime20 ne 0 then (RechTime20 / RechTimeTotal) else 0);
RechFrac30 = if (RechTime30 ne 0 then (RechTime30 / RechTimeTotal) else 0);
RechFrac40 = if (RechTime40 ne 0 then (RechTime40 / RechTimeTotal) else 0);
RechFrac50 = if (RechTime50 ne 0 then (RechTime50 / RechTimeTotal) else 0);
RechFrac60 = if (RechTime60 ne 0 then (RechTime60 / RechTimeTotal) else 0);
RechFrac70 = if (RechTime70 ne 0 then (RechTime70 / RechTimeTotal) else 0);
RechFrac80 = if (RechTime80 ne 0 then (RechTime80 / RechTimeTotal) else 0);
RechFrac90 = if (RechTime90 ne 0 then (RechTime90 / RechTimeTotal) else 0);
RechFrac100 = if (RechTime100 ne 0 then (RechTime100 div RechTimeTotal) else 0);
```

```
Recharge10 = (GRECH10 * RechFrac10);
Recharge20 = (GRECH20 * RechFrac20);
Recharge30 = (GRECH30 * RechFrac30);
Recharge40 = (GRECH40 * RechFrac40);
Recharge50 = (GRECH50 * RechFrac50);
Recharge60 = (GRECH60 * RechFrac60);
Recharge70 = (GRECH70 * RechFrac70);
Recharge80 = (GRECH80 * RechFrac80);
Recharge90 = (GRECH90 * RechFrac90);
Recharge100 = (GRECH100 * RechFrac100);         #in mm/hour

GRECH = (Recharge10+Recharge20+Recharge30+Recharge40+Recharge50+Recharge60+
Recharge70+Recharge80+Recharge90+Recharge100);              #in mm/hour

report Recharge = (GRECH * 24 * 31);                # in mm/month

##########################################################################################

#---------------Soil Erosion Model: John Thornes (1990)------------------------------

EROS = if (Landuse eq 9 then 0 else (K * (OFLOW) ** M * tan (Slopedeg) ** N * exp (-0.07 * (Vegetation))));
# in mm/month.

EROS = if (EROS lt 0 then 0 else EROS);

report Erosion = if (EROS gt Depth then Depth else EROS);          # in mm/month.

DEP = if (Landuse eq 9 then 0.0001 else (Depth - (EROS)));

report Depth = if (DEP le 0 then 0.0001 else DEP);
```

```
#################################################################################################
######################VERIFICATION OF THE MODEL:#########################################
#################################################################################################

#DIRECT OUTPUTS OF THE MODEL##############################################################
#################################################################################################

#MOISTURE SUBMODEL'S OUTPUTS#############################################################

report MoistTSS = maptotal (Moist)/ActiveCells;

report MoistureTSS = timeoutput (Lusenom, areatotal (Moist, Lusenom) div LuseArea);

#POTENTIAL INFILTRATION SUBMODEL'S OUTPUTS##############################################

#report PInfilTSS = maptotal (PotInfil)/ActiveCells;

#report PotInfilTSS = timeoutput (Lusenom, areatotal (PotInfil, Lusenom) div LuseArea);

#RUNOFF SUBMODEL'S OUTPUTS#################################################################

#report RoffTSS = maptotal (Runoff)/ActiveCells;

#report RunoffTSS = timeoutput (Lusenom, areatotal (Runoff, Lusenom) div LuseArea);

#ACTUAL INFILTRATION SUBMODEL'S OUTPUTS#################################################

#report AInfTSS = maptotal (AINF)/ActiveCells;

#report AInfilTSS = timeoutput (Lusenom, areatotal (AINF, Lusenom) div LuseArea);
```

```
#RUNOFF ROUTING SUBMODEL'S OUTPUTS###########################################################

report OflowTSS = timeoutput (Pits, OFLOW);

#POTENTIAL EVAPOTRANSPIRATION SUBMODEL'S OUTPUTS########################################

#report PETTSS = maptotal (PET)/ActiveCells;

#report PotEvapTSS = timeoutput (Lusenom, areatotal (PET, Lusenom) div LuseArea);

#ACTUAL EVAPOTRANSPIRATION SUBMODEL'S OUTPUTS###########################################

#report AETTSS = maptotal (ActET)/ActiveCells;

#report ActEvapTSS = timeoutput (Lusenom, areatotal (ActET, Lusenom) div LuseArea);

#RECHARGE SUBMODEL'S OUTPUTS###################################################################

#report RechTSS = maptotal (Recharge)/ActiveCells;

#report RechargeTSS = timeoutput (Lusenom, areatotal (Recharge, Lusenom) div LuseArea);

#SOIL EROSION SUBMODEL'S OUTPUTS################################################################

report ErosionTSS = maptotal (Erosion)/ActiveCells;

report SoilErosionTSS = timeoutput (Lusenom, areatotal (Erosion, Lusenom) div LuseArea);

#INDIRECT OUTPUTS OF THE MODEL##################################################################
###########################################################################################################
```

#VARIABLE: RFMoist (Excess of Water from Moist Model)##########################################

#report RFMoistTSS = maptotal (RFMoist)/ActiveCells;

#report RRoffMoistTSS = timeoutput (Lusenom, areatotal (RFMoist, Lusenom) div LuseArea);

#VARIABLE: Sum of Runoff without the excess of water from Moisture model###############################

#report SumRoffSS = maptotal (SumRunoff)/ActiveCells;

#report SumRunoffTSS = timeoutput (Lusenom, areatotal (SumRunoff, Lusenom) div LuseArea);

#VARIABLE: Depth#####################################################################################

#report DepthTSS = maptotal (Depth)/ActiveCells;

#report SoilDepthTSS = timeoutput (Lusenom, areatotal (Depth, Lusenom) div LuseArea);

#VARIABLE: GRECH###################################################################################

#report GRECHTSS = maptotal (GRECH)/ActiveCells;

#report GRechargeTSS = timeoutput (Lusenom, areatotal (GRECH, Lusenom) div LuseArea);

#VARIABLE: Erosion####################################################################################

#report ErosTSS = maptotal (EROS)/ActiveCells;

#report SErosTSS = timeoutput (Lusenom, areatotal (EROS, Lusenom) div LuseArea);

**THE PROCEDURE CARRIED OUT FOR THE VALIDATION OF THE DIS MODEL CONSISTED OF SEVERAL STEPS:**

1. In PCRaster and on the basis of the scalar clone map, which has a value of "1" in each cell of the overall catchment area, a new map with missing values (mv) is created through: pcrcalc Newmv.map = ln (clonescalar.map – 1). This operation computes the logarithm of "0", which by definition is infinite and is considered as a missing value in PCRaster code.

2. Selection of one of the several maps used in the DIS model (e.g. solar radiation map), and then identification of a single cell with a unique value in the overall map. Afterwards performance of: pcrcalc Newclonescalar.map = if (selected.map eq uniquevalue then 1 else Newmv.map). This operation creates a new scalar clone map with one cell with a value of "1" and the rest with missing values (mv). This unique cell is the one used for running the model (5 $m^2$ area). This new clone scalar map will be the basis for the creation of the rest of required maps.

3. In PCRaster, by typing: dir*.map > text.txt, and then notepad test.txt, all the maps in the work directory will be listed in a text file. From this file, new maps are created on the basis of the new clone scalar and the original maps (i.e. pcrcalc newslope.map = slope.map * newclonescalar.map). Resultant maps will have one single value associated to selected cell, and the rest with mv.

4. Input solar radiation maps will also be developed in the same way as the rest of maps. In total twelve maps (1 year) will be created, each one with the correspondent monthly solar radiation value obtained from the AWS.

5. Monthly rainfall and temperature time series files, needed in the model, will be created on the basis of field and AWS data, respectively.

6. Once the model script has been adjusted for a particular land use, the correspondent validation test is run.

7. Finally, comparison and assessment of soil erosion and overland flow output values, simulated by the DIS model, with those measured in the field per each month throughout the one-year study period.