UNIVERSITY OF THE ALGARVE
Faculty of Sciences and Technology

Assessing soil erosion due to land use change at the Alqueva reservoir surrounding area

Vera Lúcia Matias Ferreira

Dissertation Submitted in Fulfillment of the Requirements for the Degree of PhD in Marine, Earth and Environmental Sciences, specialization in Environmental Management.

Supervisor: Dr. Thomas Panagopoulos

Faro, 2015

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Avaliação da erosão resultante das alterações de uso do solo na área envolvente à albufeira do Alqueva

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Declaration of authorship of work

I hereby declare that, I am the author of this work, which is original and unpublished. Authors and works consulted are properly cited in the text and included in the list of references included.

Vera Lúcia Matias Ferreira

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“The highest reward for a person's toil is not what they get for it, but what they become by it.” (John Ruskin)

“Unfortunately, soils are made by nature and not by men and the products of nature are always complex.” (Karl Terzaghi)

To my parents,

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ABSTRACT

Soil erosion is one of the most dynamic environmental and economic threats in Mediterranean regions. As a consequence of water availability in the surrounding area of the Alqueva reservoir, new challenges were created. The conversion from native Montado grassland to intensive and irrigated agriculture, the development of golf resorts and the ongoing climate change were insufficiently considered for the erosion problem during the environmental impact study of the Alqueva project, and consequently there is an urgent need to delineate a sustainable land management for the region.

The main objective of this investigation was to assess current and future soil erosions in the surrounding area of the Alqueva reservoir using the Revised Universal Soil Loss Equation (RUSLE) in combination with Geographic Information Systems (GIS). Different soil erosion factors, the main causes and consequences, and also spatial variability and seasonality were investigated, and a simulation model was developed to support decision based on the acquired scientific knowledge.

On the first part of the study, the RUSLE equation was applied at field scale, and different land uses were selected for erosion assessment (Montado grassland, lucerne cultivation, olive orchard and vineyard). The spatial variability analysis (with geostatistics and HJ-Biplot) indicates that the intensification of land use, with tillage practices and vegetation removal, is likely to increase the susceptibility to soil erosion (soil erodibility). The effect of seasonality on soil erosion was confirmed, with the autumn season contributing the most to annual soil erosion (around 65%).

Future soil erosion scenarios were investigated for the entire study area, according to the expected land use changes (which affect vegetation cover) and climate changes (which affect rainfall-runoff erosivity). The forecasting scenarios of land use changes indicated that the intensive agriculture area is likely to increase, as well as sparse and xerophytic vegetation and rainfall-runoff erosivity. As a consequence, soil erosion in the study area is forecasted to increase from 1.78 t/ha to 3.65 t/ha by 2100. A backcasting scenario was investigated by considering the application of soil conservation practices, that will decrease soil erosion considerably to an average of 2.27 t/ha. For each scenario studied, the sediment delivery was assessed, and for the worst case scenario in 2100, an annual sedimentation value of 182 000 tonnes is predicted for the study area.

Finally it was developed a dynamic simulation model for soil erosion performed on Stella, and a graphic user interface as a decision support tool allowing the user (*e.g.* decision maker) to create, modify, save, and select site specific data. The system simulates the risk for soil erosion for particular local characteristics and land use, and then suggesting soil conservation practices to decrease susceptibility to erosion.

In conclusion, due to its characteristics, the study area is very vulnerable to land degradation processes, which is expected to worsen in the future. The distribution maps provide for a better understand of soil erosion and its processes under local conditions, and for the identification of critical periods, high-risk areas, and their respective causes. This information is crucial to delineate local strategies for sustainable land management, and future scenarios reveal the importance of considering the effects of land use and climate change. The decision support system is a useful tool for the exchange of scientific knowledge; however, close collaboration between scientists and local stakeholders is essential to preserve the natural resources and avoid unnecessary costs. In future research, collaboration with international projects will be important to exchange information and knowledge as a key element in the global effort to fight land degradation and to promote sustainable land management.

Keywords: Soil erosion, Alqueva reservoir, RUSLE, GIS, Land-use/cover change, Climate change, Seasonality, Spatial variability, Sedimentation, Decision support, Sustainable land management.

RESUMO

A erosão do solo é um dos mais dinâmicos problemas ambientais e económicos nas regiões mediterrânicas. Como consequência da disponibilidade de água, a zona envolvente da albufeira do Alqueva apresenta agora novos desafios e têm-se verificado alterações na paisagem local resultantes sobretudo do aumento do cultivo intensivo em regadio. Além disso, prevê-se um aumento de projetos turísticos (com áreas de golfe) e a produção de biomassa para bioenergia. Estas alterações do uso do solo, associadas às expectáveis alterações climáticas, podem intensificar a erosão do solo na região, a qual terá impactos não só a nível da sustentabilidade e produtividade do solo, mas também no aumento da sedimentação e degradação da qualidade da água da albufeira do Alqueva.

Face a esta problemática, e para a definição de uma estratégia de conservação do solo na região, era urgente avaliar os efeitos dessas alterações na erosão do solo, questão à qual foi dada pouca significância aquando da avaliação de impacte ambiental do projeto Alqueva. Este foi o principal objetivo desta investigação, e envolveu o uso da RUSLE (Revised Universal Soil Loss Equation) em combinação com Sistemas de Informação Geográfica (SIG) para avaliar a atual e futura erosão do solo na área envolvente à albufeira do Alqueva, compreender quais as principais causas e efeitos, bem como variações espaciais e temporais da mesma. Outro objetivo da tese era criar uma ferramenta de apoio à decisão que sustentasse a gestão e planeamento do uso do solo na região.

A investigação foi assim dividida em três partes. A primeira parte envolve a aplicação da RUSLE em áreas experimentais (Herdades) selecionadas, para obtenção de mapas de distribuição da erosão do solo e dos respetivos fatores, incluindo uma análise da variabilidade espacial das propriedades do solo e correlações em diferentes usos do solo, efetuada com recurso a técnicas de geostatística e HJ-Biplot. Uma análise detalhada da sazonalidade de alguns fatores e a sua preponderância na erosão anual do solo é igualmente inserida nesta fase. Na segunda parte é efetuada a simulação de futuros cenários de erosão do solo, para toda a área de estudo, de acordo com as expectáveis alterações climáticas e modificações do uso do solo, sendo para isso utilizada uma abordagem "forecasting" e "backcasting". Por fim, é apresentada um sistema de simulação da erosão do solo construída com base na informação adquirida e com objetivo de apoiar a tomada de decisão na região.

As áreas experimentais foram selecionadas de forma a incluir diferentes usos do solo, nomeadamente o uso tradicional (montado) e outros usos com recurso à rega (cultivo de luzerna, olival e vinha). Os resultados indicam que a intensificação do cultivo com rega, leva a um aumento da suscetibilidade à erosão do solo (erodibilidade do solo), consequência principal das frequentes mobilizações do solo e remoções da vegetação, Confirma-se através dos mapas de distribuição, a variabilidade sazonal da erosão do solo relacionada com as alterações na cobertura pela vegetação e variação da intensidade das precipitações ao longo do ano. Existe uma maior vulnerabilidade à erosão no outono, quando a erosividade das chuvas atinge o seu máximo e a vegetação (especialmente em sistemas naturalizados) é ainda muito baixa após um longo período quente e seco como o verão. Verifica-se que a erosão durante esta estação pode contribuir com cerca de 65% para a erosão anual.

Após a análise dos futuros cenários de alteração de uso do solo na área de estudo, verifica-se que há uma tendência para o aumento da agricultura de regadio e como consequência das alterações climáticas, um aumento da vegetação esparsa e xerofítica (que afeta a cobertura pela vegetação) e a intensificação da erosividade da precipitação. Como consequência das alterações uso do solo e do clima, é estimado um aumento da erosão do solo de 1.78 t/ha em 2006 para 3.65t/ha em 2100. De acordo com o cenário "backcasting" que considera a implementação de práticas de conservação do solo, apesar das alterações do uso do solo e do clima, espera-se uma diminuição da erosão para 2.27 t/ha (cerca de 38%). Para cada cenário estudado foram analisados os valores de sedimentação, nomeadamente a quantidade de sedimentos que efetivamente escoam na albufeira, e estima-se que para o pior cenário (em 2100) o valor anual seja cerca de 182 000 toneladas.

O sistema dinâmico de simulação da erosão é criado de forma a apoiar a gestão do uso do solo na área de estudo. Este sistema inclui um modelo empírico da erosão do solo criado através do programa Stella, e uma interface criada para que o utilizador (*e.g.* gestor na região) ao inserir as características locais ou simplesmente selecionando uma área no mapa, consiga uma simulação da suscetibilidade de um determinado solo à erosão. De acordo com o uso e gestão estabelecidos, e a simulação efetuada, são sugeridas recomendações de práticas sustentáveis para prevenção da erosão do solo.

Através dos resultados desta investigação conclui-se que é indispensável a implementação de uma estratégia sustentável de gestão e conservação do solo na área envolvente à

albufeira do Alqueva. Devido às características da região existe um elevado risco de erosão do solo e o mesmo é expectável de agravar face às alterações climáticas e de uso do solo na região. Os mapas de distribuição e todos os cenários estudados são uteis não só na identificação de períodos mais críticos para a erosão do solo, mas também no reconhecimento de áreas de elevada vulnerabilidade, e as principais causas e consequências de determinadas modificações. Os futuros cenários revelam a importância da definição de uma estratégia baseada não só nas características locais e padrões de sazonalidade, mas também considerando os efeitos das alterações climáticas e de uso do solo. Uma ferramenta de apoio à decisão, como o modelo criado, é útil na partilha do conhecimento científico com os gestores e decisores da região. No entanto, a estreita colaboração entre cientistas e agentes locais é essencial de forma a salvaguardar os recursos naturais da região e evitar custos desnecessários associados. Como futura pesquisa, a colaboração com projetos internacionais será determinante na partilha de conhecimento e estratégias utilizadas fundamentalmente em áreas com problemática semelhante (região mediterrânica).

Palavras-chave: Erosão do solo, Albufeira do Alqueva, RUSLE, SIG, Alterações de uso do solo, Alterações climáticas, Sazonalidade, Variabilidade espacial, Sedimentação, Apoio à decisão, Conservação do solo, Sustentabilidade.

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1. INTRODUCTION

1.1. SOIL EROSION THREAT

Undoubtedly, the soil is one of the most important environmental components, and yet it remains an ill-treated and undervalued natural resource, associated with its multi-functionality and inadequate specific legislation. A subsequent threat to this natural resource is the intensification of soil erosion, a complex and dynamic process characterized by the deterioration of soil quality and productivity. Soil erosion begins with the increase of runoff and the loss of topsoil, reducing water and nutrient storage capacity (Yang *et al.*, 2003; Pimentel, 2006; Li *et al.*, 2009; Hancock *et al.*, 2015). In addition to the impact on fertile land, soil erosion affects water systems, the health of ecosystems, and food provision. There are off-site negative impacts associated with the increase of runoff that can transport sediments into rivers and reservoirs, causing pollution and reducing their lifetime (Pandey *et al.*, 2007; Ludwig *et al.*, 2009; Haregeweyn *et al.*, 2013). As the world population grows in number, soil erosion becomes more severe, bearing direct consequences in quality of life, and increasing concerns regarding soil conservation (Troeh *et al.*, 2003).

The rhythm of worldwide land use changes and landscape structure modifications have been associated with the loss of biodiversity, depletion of natural resources, and intensification of soil erosion. Some researchers (Kosmas *et al.*, 1997; Islam and Weil, 2000; Yang *et al.*, 2003; Blavet *et al.*, 2009; Cerdà *et al.*, 2009; Cantón *et al.*, 2011; Leh *et al.*, 2013; Wang and Shao, 2013; Salvati and Colantoni, 2015) have demonstrated that alterations on vegetation cover and/or farm procedures greatly affect the properties of soil and hydrological processes. In general, cultivated lands show the highest erosion yield (Erskine *et al.*, 2002; Garcıía-Ruiz and Lana-Renault, 2011).

The Mediterranean regions are particularly prone to this phenomenon because they have been subjected to the anthropogenic pressure, due to changes in agricultural technologies and social, political and economic development (Bakker *et al.*, 2008), combined with specific seasonal conditions (Karydas *et al.*, 2009; García-Ruiz *et al.*, 2013). According to Grimm *et al.* (2001) almost one third of Portugal is in high risk of erosion. Furthermore, the expected climate change will multiply the forces responsible for soil erosion, increasing the susceptibility of populations and their environments (Lal *et al.*, 2011).

1.2. SOIL EROSION ASSESSMENT TOOLS

Over the past few decades, numerous advances have been made to assess soil erosion, in order to overcome the costs and unfeasibility of monitoring *in situ*. Modelling soil erosion can provide a quantitative and consistent estimation of the phenomenon under various conditions. There is not a most suitable model for all applications. Substantial investigation into soil loss modelling has taken place, and empirical, conceptual and physical based models have been put into practice and applied in specific environments. The most appropriate model depends on a number of factors, such as: the intended use and the characteristics of the catchment considered; the data requirements; the capability, accuracy, validity and reliability of the model; the objectives of the model users; the ease of use of it; the scales at which model outputs are required; and the hardware requirements (Merrit *et al.*, 2003; Bhattarai and Dutta, 2008; Volk *et al.*, 2010). The physics-based models are the most complex, for they are based on physical equations and require and require highly detailed information (*e.g.* PESERA, Kirkby *et al.*, 2004; WEPP, Flanagan and Nearing, 1995; EUROSEM, Morgan *et al.*, 1998). The conceptual models play an intermediary role and usually include a generalized and aggregated description of catchment processes (*e.g.* AGNPS, Young *et al.*, 1989; SWAT, Arnold *et al.*, 1998). The empirical models are generally the simplest, based on stochastic relationships which are limited to the conditions for which they have been developed (*e.g.* USLE, Wischmeier and Smith 1978; RUSLE, Renard *et al.*, 1997; USPED, Mitasova *et al.*, 1996).

The empirical Universal Soil Loss Equation (USLE) and the Revised Universal Soil Loss Equation (RUSLE), are the most widely used models for estimating annual soil loss, becoming the standard technique for soil conservation (Morgan, 2005). USLE was firstly developed to estimate soil loss from specific agriculture fields in the United States of America (USA). The RUSLE was developed with the goal of taking advantage of knowledge and new data obtained, retaining the same factors but changing the way some factors are determined. Furthermore, it allows for the estimation of different crops and management systems (Kinnel, 2010). Significant RUSLE/USLE disadvantages consist of the overestimation of erosive slope lengths and the lack of possibilities for process-oriented simulations such as sediment transport (Winchell *et al.*, 2008). However, RUSLE still possesses the highest benefit in terms of applicability (less input data requirements) and reliability of soil loss estimates (Desmet and Gooverts, 1996; Ferro, 2010).

Due to the spatial variation in ecosystems, the erosion models often require moderate to elevated amounts of spatial data such as topography, soil and land use, which can be effectively handled through Geographic Information Systems (GIS) (Bhattarai and Dutta, 2007; Mulligan, 2004; Wachal and Banks, 2007). The GIS can be used for the discretization of the catchment/watershed into small grid cells (Bhattarai and Dutta, 2007). GIS is an integrated suite of computer-based technologies and methodologies that link geographic information with descriptive information on different themes and from different sources (Burrough and McDonnell, 1998). It includes a powerful set of tools for collecting, storing, retrieving at will, transforming, analysing and presenting spatial data from the real world for a particular set of purposes (Burrough and McDonnell, 1998; Davis, 2001). The GIS has been used more and more in combination with soil erosion models, because it makes spatial distribution of soil erosion estimations feasible, and it allows the representation of different scenarios from various changing land use conditions and management alternatives in space and time (Fu *et al.*, 2006; Shi *et al.*, 2004; Bühlmann, 2006; Bhattarai and Dutta, 2007; Terranova, 2009; Prasannakumar *et al.*, 2011). Also, the GIS allows for the use of Geostatistics (sub-section 2.3.2).

1.3. SEASONALITY OF SOIL EROSION

Because soil erosion is the result of many processes and interactions, it is also time-variant. There is evidence in literature that soil erosion varies not only between years but also throughout the year (between seasons), especially due to intra-annual rainfall and variations in vegetation cover (Van Leeuwen and Sammons, 2003; Lu *et al.*, 2003; Grazhdani and Shumka, 2007; Panagos *et al.*, 2011). These rainfall-runoff and vegetation dynamics are particularly investigated in the Mediterranean regions (Cerdà, 2002; López-Bermudez *et al.*, 1998; Boardman *et al.*, 2009; Diodato and Bellocchi, 2010; Alexandridis *et al.*, 2013), and their knowledge is essential to successfully delineate specific soil conservation strategies and sustainable land management throughout the year.

Rainfall-runoff erosivity is determined by the climate region, the seasonal pattern of rainfall and the occurrence of storms (Evrard *et al.*, 2010). In the Mediterranean area, rainfall has great temporal variability over the years and particularly throughout the year, resulting in a strong seasonality of erosion amounts (Renschler *et al.*, 1999; Cerdà, 2002; González- Hidalgo *et al.*, 2007; Regüés and Gallart, 2004; Lana-Renault *et al.*, 2007; Boardman *et al.*, 2009; Diodato and Bellocchi, 2010).

Vegetation cover is an important factor in protecting the soil against erosion, and the efficiency varies greatly between vegetation types, which are always related to patterns in land use (Zhou *et al.*, 2006). This RUSLE factor is frequently derived from satellite images; however, because of the lack of data, very few images (or only one image) are used each year, despite being temporally variable according to plant phenology (Alexandridis *et al.*, 2013). Vegetation cover usually presents temporal dynamics due to differences in availability of water in the soil and in temperatures, and it is often induced by management practices. Some investigations have focused on these seasonal changes of vegetation cover (López-Bermudez *et al.*, 1998; Camacho-De Coca *et al.*, 2004; Gallo *et al.*, 2005).

1.4. SUSTAINABLE LAND MANAGEMENT AND DECISION SUPPORT SYSTEM

Soil erosion is a natural process, but it can be reduced to a maximum acceptable level or soil loss tolerance. Because of this, there is a challenge to promptly and efficiently study ecosystem changes, analyze its impacts, and proactively support sustainable land management (SLM) (Schwilch, 2012a).

Land planners and land owners are responsible for soil management actions, but they are often unaware of their role in the larger-scale erosion process. They need support from researches to help them effectively evaluate the social, economic and environmental consequences of alternative management scenarios and to specify the viable SLM strategies for soil erosion control in the region. According to Peterson *et al.* (2003), decisions based on simulated scenarios provide better flexibility when facing irreducible uncertainty.

Appropriate instruments are required for the exchange of knowledge. Decision support systems (DSS's) have been created to improve the decision-making process for sustainable land management and to help developing dynamic soil conservation plans

(Pertiwi *et al.*, 1998; Bathurst *et al.*, 2003; Dragan *et al.*, 2003; Manos *et al.*, 2010). A DSS has been defined in many different ways, but can be regarded in general as an interactive computer-based system that helps people use computer communications, data, documents, knowledge and models to solve problems and make decisions (Matthies *et al.*, 2007). Some environment DSS's allow for the simulation of future alternative scenarios and can suggest alternative options for decision-makers (Manos *et al.*, 2010; Panagopoulos *et al.*, 2014).

1.5. STUDY RATIONALE AND OBJECTIVES

The management of reservoirs is of major importance when it comes to water supply in Portugal, not only for consumption but for energy proposes as well. The construction of the Alqueva dam in 1998, whose main propose was the creation of a water reserve in the Alentejo region, has generated the largest artificial reservoir in Portugal and in Western Europe.

Erosion has a particular influence when dams are built. Dams trap the incoming sediment in their reservoir while also changing the natural stream flow and sediment load downstream. Furthermore, an increased rate of erosion in the surrounding area can lead to problems related to the decrease in the reservoir's life time (Julien, 2010).

With the construction of the Alqueva dam, its surroundings have faced new challenges, and land use changes have taken place as a consequence of water availability. Some of these land use changes include the conversion from the native Montado grassland to intensive agricultural uses with irrigation. Furthermore, several developments were projected around the large lake in order to build rural tourism and golf resorts, and there are plans to produce biomass for bioenergy.

During the impacts assessment of the Alqueva dam, insufficient attention was paid to soil erosion problem despite the challenges created with the project. Consequently, no adequate soil conservation plan was delineated for the region in order to protect the resource soil and the reservoir. The increase of sediment transport in the surrounding area means more sedimentation in the reservoir, which is an environmental and economic threat, for it can create water quality problems and decrease financial benefit. In sum, soil erosion has to be carefully evaluated to adequately delineate a sustainable land management strategy.

Different projects at large scales have been created with the contribution of the European Commission, in which the risk of soil erosion was estimated for many European countries, including Portugal (*e.g.*, PESERA, Kirkby *et al.*, 2004) and vulnerable areas that were identified in the Mértola municipality (Alentejo) (*e.g.*, MEDALUS, Kosmas *et al.*, 1999). Furthermore, actual soil loss was measured for a period of 44 years at plots of approximately 166.7 m^2, at the Vale Formoso experimental center (Roxo and Casimiro, 2004).

Nevertheless, the knowledge regarding soil erosion rates and processes at a smaller scale, namely on the surrounding area of the Alqueva reservoir, considering different regional directions and conditions, is essential in order to promote soil conservation, ecosystem sustainability, and to protect the reservoir. Facing that, the main goals of the present research were:

- To predict soil erosion rates under current cropping conditions using the RUSLE modelling combined with GIS tools;
- To study erosion processes and each factor considered by RUSLE, and their spatial distribution;
- To investigate the effect of different land uses on soil erosion;
- To apply geostatistic and HJ-biplot tools for a better spatial variability analysis of soil properties, finding connections with management practices and erosion processes;
- To investigate seasonality of soil erosion due to intra-annual variations in rainfall-runoff and vegetation cover;
- To construct future soil erosion scenarios, accounting the land use trajectories in the region and climate change;
- To develop desirable scenarios to decrease soil loss by applying local sustainable land management (SLM) innovations in specific fields at the study area;
- To develop a dynamic simulation model of soil erosion connected to a decision support system (DSS).

1.6. THESIS OUTLINE

This thesis is divided in four main chapters following a typical outline, namely the introduction, the methodology, the results/discussion, and the conclusion. Chapters are divided into additional subchapters.

The general introduction was presented previously in this chapter (Chapter 1), allowing the familiarization with the main themes of this thesis, and the central motivations and objectives of the investigation.

Chapter 2 corresponds to the methodology. The first section characterizes the study area, describing the Alqueva project, providing details about the surrounding reservoir area and specific aspects about the Alentejo region, and presenting experimental study land uses. In the second section it reviews and discusses the methodologies used to investigate soil erosion by RUSLE, giving calculation details about each factor and its estimation using the GIS. An approach about spatial variability analysis used in the study was explained in section 2.3, mentioning each of the methods used, namely normal statistics, geostatistics and HJ-Biplot. Section 2.4 presents the methodology used to study the future scenarios of soil erosion for the entire surrounding Alqueva area and the Section 2.5 explains the method used to estimate the sediment yield and amount values associated to these scenarios, to better realize the significance of soil erosion. The last section of this chapter presents the methodology used to create a simple and alternative simulation model connected to a DSS to be used by planners and decision makers in the region.

The results obtained with the methodology described in Chapter 2 are presented and debated in Chapter 3. Furthermore, the results from a specific analysis of soil erosion seasonality are accessible in the sub-section 3.1.6.

Conclusions about research questions, limitations and considerations about future research and possible trends are presented in the last chapter (Chapter 4).

2. METHODOLOGY

2.1. STUDY AREA

2.1.1. Alqueva Project

The study area is located in the south-central part of Portugal, a region called the Alentejo, known as a semi-arid area experiencing a long term process of biophysical and human desertification. In the 50s, the idea of unproductivity in that region was related to the predominance of non-irrigated agriculture, and the solution plan was the construction of the Alqueva dam (7º30' W, 38º15' N), on the Guadiana river (Sanches and Pedro, 2007).

The pros and cons were discussed for decades and the Alqueva dam project began in 1998, and was finished by 2002, when the dam's floodgates were shut. On the 12th of January 2010, the lake was complete to its planned level, generating the largest artificial reservoir in Western Europe. The reservoir has a surface area of 250 km^2 (from which 35 km^2 is in Spain), a total capacity of 4.15 km^3, a total shoreline of approximately 1100 km, and is 83 km in length (EDIA, 2012).

The Alqueva reservoir is the main water source of a strategic multi-propose system (Alqueva Multi-purpose Undertaking - EFMA), which is a responsibility of EDIA – Empresa de Desenvolvimento e Infra-estruturas do Alqueva, SA. The main objectives of the strategic project were (Sanches and Pedro, 2007):

- Creating a water reserve for agriculture;
- Ensuring the water consumption for nearby populations;
- Producing electric energy;
- Developing tourism and leisure activities;
- Promoting the region's economic development and fight the effects of desertification.

The Alqueva Project is a system that ensures the transport of water, consisting of 68 reservoirs and dams, 300 km of primary network - which connect the main dams of the system -, 1577 km of buried pipelines on the secondary network, 52 pumping stations and 5 mini hydro plants (EDIA, 2012). It is through the Alqueva reservoir that other dams interconnect to ensure water supply even in periods of extreme drought to an area of around 10,000 km^2 in a total of 20 municipalities in the districts of Beja, Évora, Portalegre and Setúbal (200 000 inhabitants). Its hydroelectricity infrastructures produce enough energy to supply a city with more than 500 000 inhabitants (EDIA, 2012).

The Alqueva general irrigation system is projected to serve an area of about 120,000 ha and is divided into three subsystems based on the water's origin: Alqueva, Pedrogão and Ardila. Almost all of the irrigation area is projected downstream (about 92%) from the Alqueva dam or in the neighborhood watershed, and around 8% (about 10 000) in the upstream area.

2.1.2. Surrounding Area of Alqueva Reservoir

Beside the irrigation area defined by the EDIA (in the Alqueva project), the adjacent upstream area has an increasing number of irrigated cultures, not accounted for in this project, due to the water being pumped directly from the Alqueva Lake or other small reservoirs. According to Ilhéu *et al.* (2011), in 2011 about 5400 ha (declared) of surrounding areas were irrigated through direct pumping. However, the irrigated area is expected to increase in the upstream surrounding area of the Alqueva reservoir. Additionally, most of the tourism projects (such as golf areas) have been projected for the surrounding area (Ferreira and Panagopoulos, 2012; Espada, 2011).

Since the objective of the investigation is the assessment of soil erosion and its effects (direct runoff and sedimentation into the reservoir) particularly due to land use changes derived from the implementation of the Alqueva project, the study area involves the adjacent surrounding area of the Alqueva reservoir, presented on the Figure 1. This area was defined according to the limits of landscape management plan specifically focused on this territory, namely the Regional Plan for the Surroundings of Alqueva lake (PROZEA) (CCDRA, 2001). The PROZEA delineates some strategies and potentialities for agriculture, tourism, and also sustainable and urban development in the region, which are then used to create future land-use/cover change scenarios (sub-section 2.4.1.1).

Excluding the area submerged by the lake, the research area integrates six municipalities of the Alentejo region: Alandroal, Barrancos, Moura, Mourão, Portel and Reguengos de Monsaraz (Figure 1 and Table 1). The total study area corresponds to 300 270 ha (as shown in Table 1), including the reservoir.

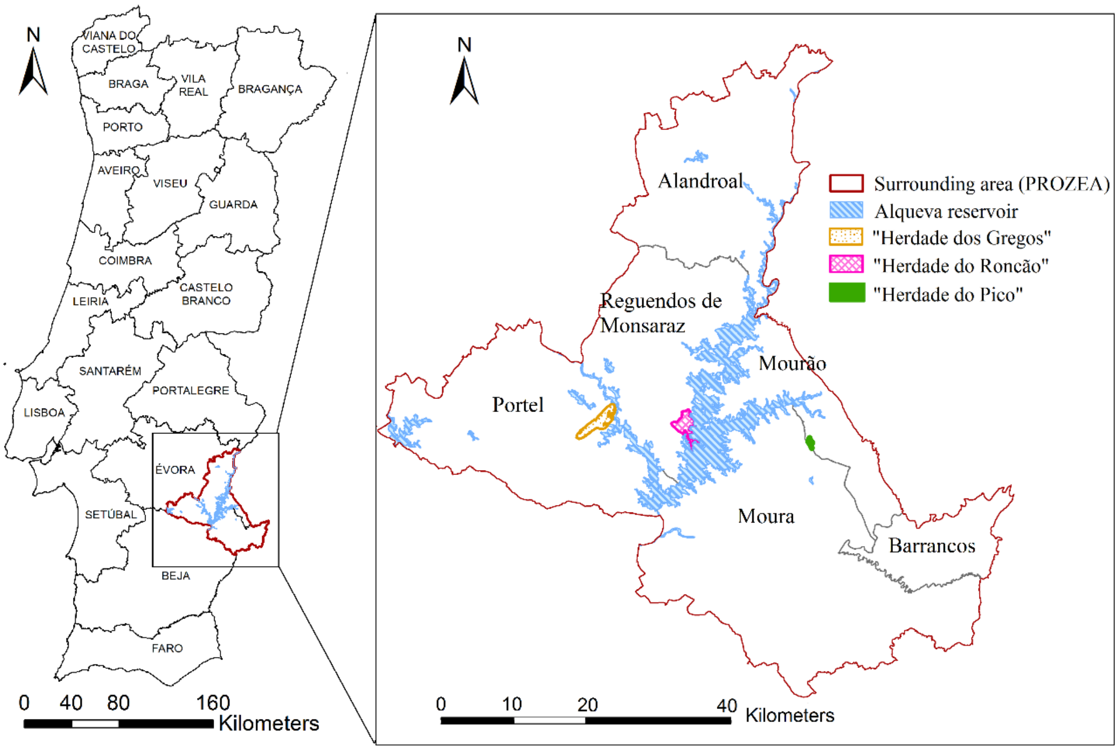

Figure 1 – Study area location.

Table 1 - Total extent for the study area and for each municipality (in hectares).

Municipality	Area (ha)
Alandroal	54480
Barrancos	16870
Moura	95250
Mourão	27770
Portel	60340
Reguengos de Monsaraz	46560
Total (study area)	300270

The study area region is demographically characterized by low population densities and has experienced a decline of its population due to rural exodus. The total Alentejo region (which corresponds to the districts of Beja, Évora, Portalegre and part of Setúbal, a total of 2 727 600 ha) lost 25% of its population between 1950 and 1970. Currently, the population loss continues at a 5% rate for the 2001-2011, with a population density of 18.5 hab/km^2 for 2011 (INE, 2014).

The climate is continental Mediterranean (type Csa, according to Koppen classification), with mild winters and very hot and dry summer. The average temperature ranges from 24 to 28°C in hot months (July/August), and from 8 to 11°C in cold months

(December/January). The average annual precipitation for the last 30 years is approximately 500 mm. The region experiences long dry periods, given that 80% of the precipitation occurs from October through April (Ferreira and Panagopoulos, 2014).

The soils are mainly leptosols and luvissols as shown on the Figure 2 (CNA/SROA, 1978). These soil are usually characterized by low percentage of organic matter. The landscape is characterized by its hilly topography with significant altitude variations (mainly between 50 and 570 meters), as shown on Figure 3.

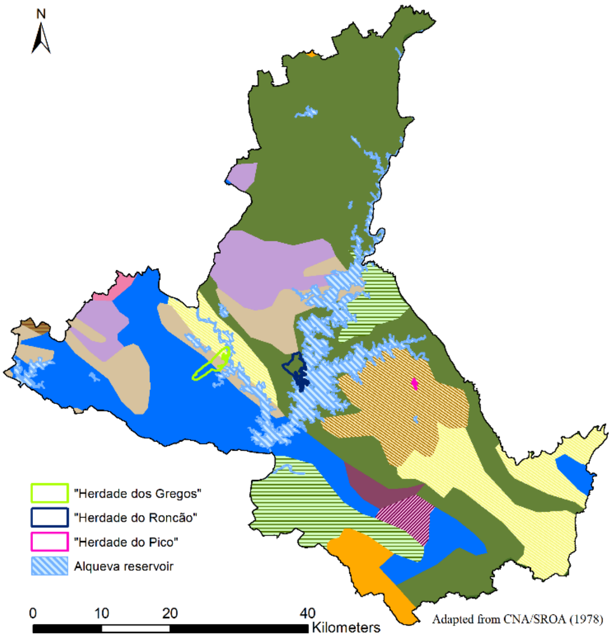

Figure 2 – Soil classification in the study area (adapted from CNA/SROA (1978)).

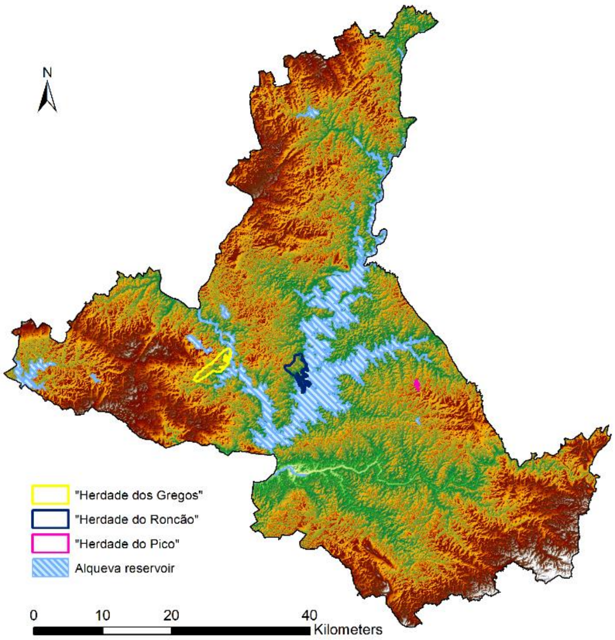

Figure 3 – Elevation values in the study area.

Before the Alqueva dam implementation, the landscape was characterized by dryness and immensity, reflecting the predominance of non-irrigated cultures, olive groves, vineyards and a typical agro-silvo-pastoral system called “Montado” (Portugal) and “Dehesa” (Spain). The traditional “Montado” is comprised of low density woodlands of cork oak (Quercus suber) and holm oak (Quercus ilex), combined with a rotation of crops/fallow/pastures (Borges *et al.*, 2010). In some montado areas, oaks are mixed with olive trees. According to the CCDRA (2001), before the Alqueva project, the Montado grasslands occupied about 54.4% of the surrounding area (study area), the olive groves around 12.6% and the vineyards less than 1%.

In the beginning of the 20th century, there was an intensification of agriculture for cereal production combined with extensive livestock breeding. This intensification led to numerous environmental impacts particularly increased soil erosion. Especially since Portugal joined the European Community in 1986, due to socioeconomic aspects presented by Jones *et al.* (2011), the abandonment of agricultural land has increased with the transition of some “Montado” systems to silvo-pastoral or total forestry systems. This change left the ecosystems more vulnerable to fires that increased its susceptibility to soil erosion (Jones *et al.*, 2011); however, in some areas it resulted in the decrease of soil erosion (Bakker *et al.*, 2008). Nowadays, the Alqueva landscape has been rapidly changing as a consequence of water availability, inducing land use change dynamics due to intensification of irrigated farming, despite the restricted area with potential for intensive agriculture (12.5%) (CCDRA, 2001). Furthermore, several tourism developments were projected around the large lake in order to build rural tourism and golf resorts, and there are plans for biomass production for bioenergy as well. Additionally to land use changes, the ongoing climate change is expected to modify vegetation patterns in this region.

2.1.3. Experimental Land Uses

In order to investigate soil erosion, using the RUSLE model in different types of land use, three experimental areas (“herdades” in Portuguese) were selected: “Herdade do Roncão”, “Herdade dos Gregos” and “Herdade do Pico”. As exposed in Table 2, four different land uses were identified in these areas: montado grassland, vineyard, lucerne cultivation, and olive tree orchard.

Table 2 - Experimental study sites, respective land uses and areas (in hectares).

Experimental Site	Land use	Area (ha)
"Herdade dos Gregos"	Montado	20.7
	Lucerne cultivation	33.5
	Olive tree orchard	57.5
"Herdade do Roncão"	Montado	782
"Herdade do Pico"	Vineyard	30

The "Herdade dos Gregos", shown in Figure 4, is located in the west part beside the Alqueva reservoir (Figure 1), in the Portel municipality. The landscape is characterized by its hilly topography with significant altitude variations (mainly between 100 and 250 meters) (Figure 4). According to the World Reference Base for Soil Resources (CNA/SROA, 1978), the types of soil in this area are: ferric luvisols (LVf), chromic luvisol (LVch) and eutric leptosols (LPeu) (Figure 2). Direct pumping from Alqueva reservoir occurs in this private property since it is near the reservoir. This farm was selected to include a diversity of land uses, including:

- A native Montado grassland (20.7 ha) characterized by a silvo-pastoral system with low density holm oaks, used as a permanent pasture for the cattle. This small area is located in the high altitudes of the "Herdade dos Gregos" (from 200 to 240 m) with a slope that varies from 1.4 to 20.9 %. Tillage (at depths of about 15 cm) took place only once every 10 years in order to decrease shrub competition, and the soil is not subjected to any fertilizer. Four years prior to the investigation, there was a fire on this farm.

- An intensive cultivation of Lucerne (33.5 ha) with irrigation (Pivot Sprinkler Irrigation System), tillage and fertilization. Direct pumping from the Alqueva reservoir takes place in this private property since it is near the reservoir. Lucerne (*Medicago sativa*) is sown four times a year and once dried is nutritional for cattle, and it incorporates nitrogen in the soil. Conventional tillage is used, involving multiple aspects: plough (at about 20 cm of depth) in the fall, fallowing cultivator (depths of about 15 cm) and disc harrow (10 cm depths) after soil tillage. Inorganic fertilizers were applied to the cultivated field at a rate of 100 kg ha^{-1} NPK. This land use is placed in the midland (194-220 m), and the slope varies from 0 to 9%.

- An olive tree orchard (57.5 ha) with irrigation, which is done in strips. This cultivation has a drip irrigation system, is fertilized once every two years and is ploughed once a year to decrease weed competition. The Olive orchard is located in the low elevations of the farm (150-186 m) and the slope varies from 0 to 14.2%.

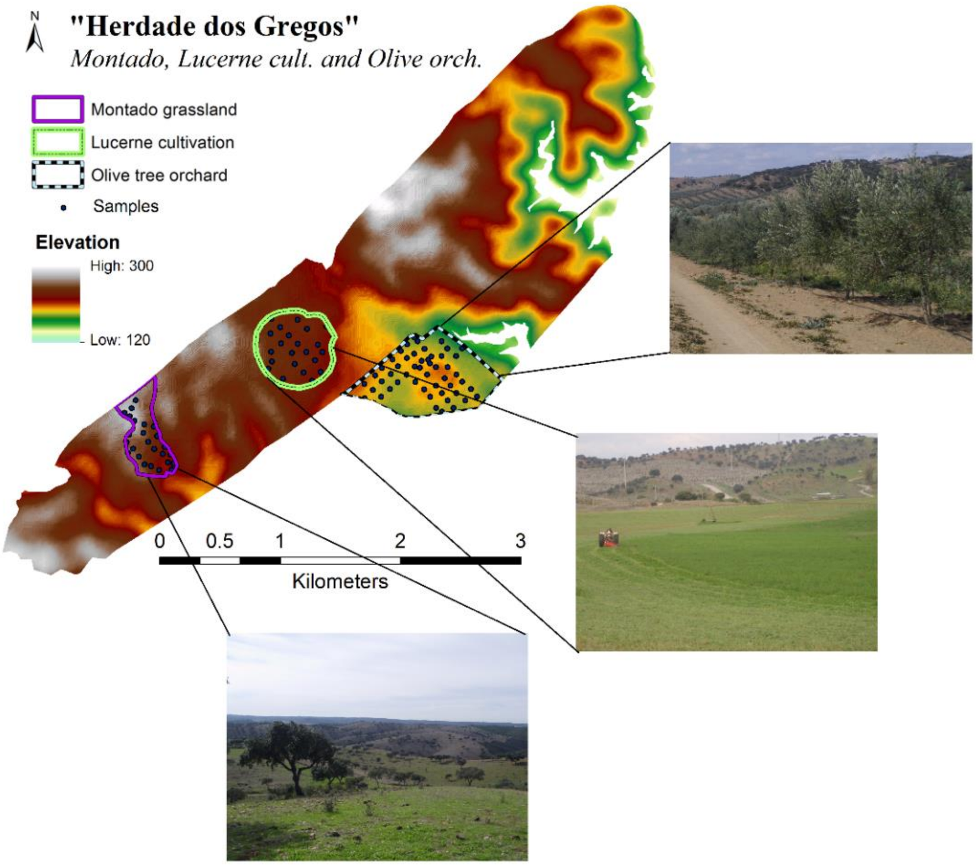

Figure 4 – “Herdade dos Gregos” experimental area.

“Herdade do Roncão”, represented in the Figure 5, is comprised of 739 ha and lies beside the reservoir in the west part, near Regengos de Monsaraz (Figure 1). The area includes a typical “Montado” characterized by a silvo-pastoral system with low density holm oaks and some olive trees; farming in this area ceased about 6 years ago. A tourism project (with golf areas, hotel and marina) is being created for this site, thus taking advantage of the attractive landscape and water availability. The altitudes in this area range between 136-215 meters and the slope from 0 to 30%, with an average of 7.5%. The soil is Eutric leptosol (LPeu) (CNA/SROA, 1978) (Figure 2).

The “Herdade do Pico” (30 ha), shown in the Figure 6, consists of only one land use, a vineyard with irrigation, ploughed between lines and fertilized. This farm is located in the eastern part of the Alqueva reservoir, in the Mourão municipality (Figure 1), and it is situated in the high altitudes (203-238 m) with a slope ranging from 0 to 17%. The soil in this area is Gleyic Luvisol (LVgl) (CNA/SROA, 1978) (Figure 2).

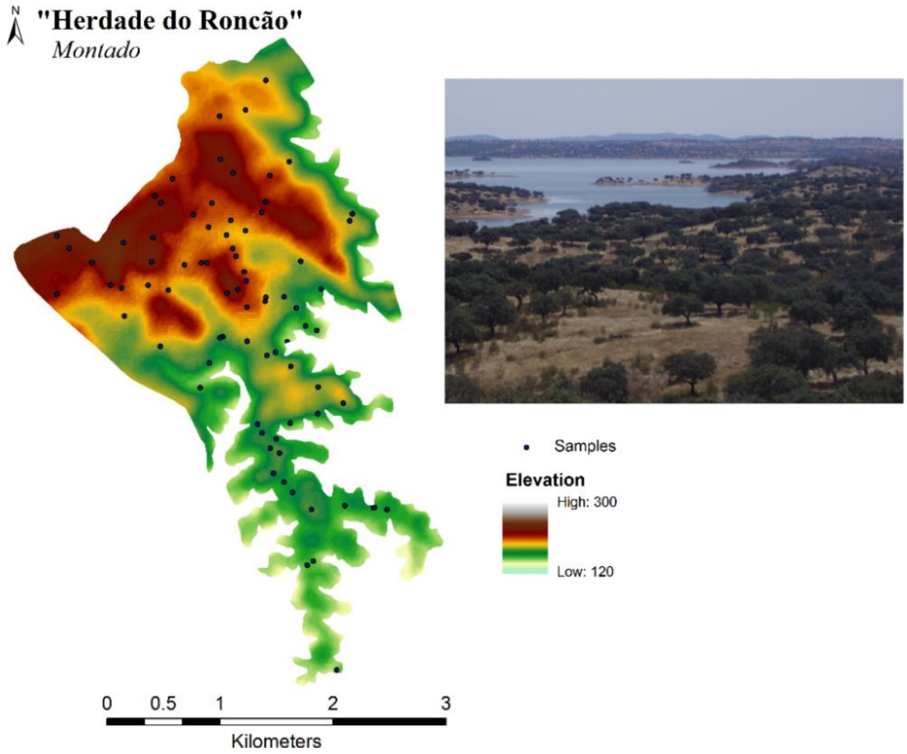

Figure 5 – "Herdade do Roncão" experimental area.

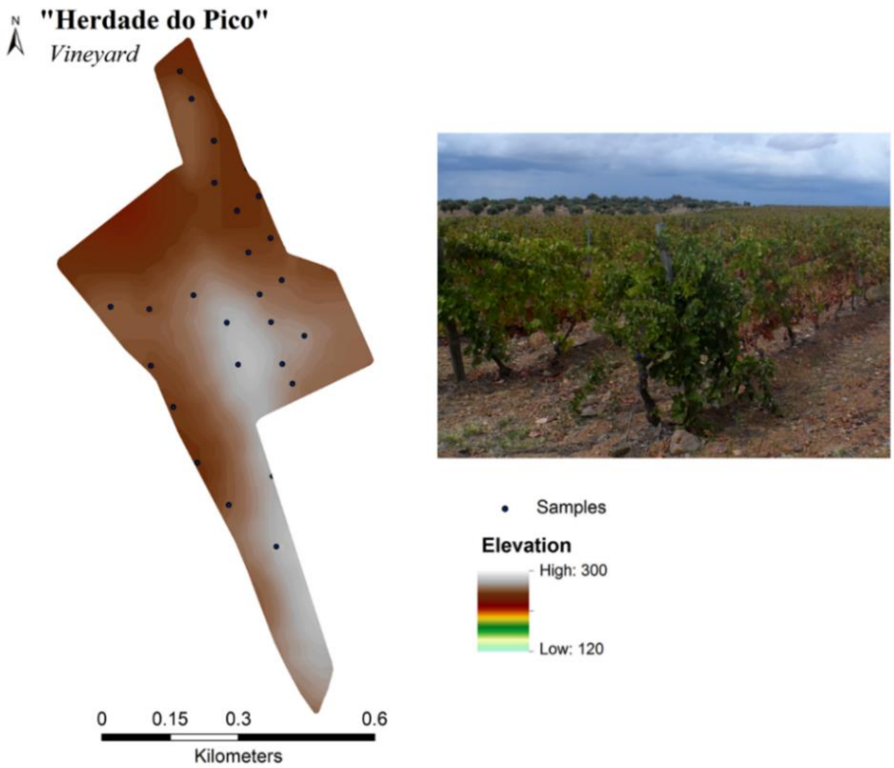

Figure 6 – "Herdade do Pico" experimental area.

2.2. SOIL EROSION BY RUSLE

The model structure of the Revised Universal Soil Loss Equation (RUSLE) by Renard *et al.* (1997) was applied as a basis for soil loss computation in this study. The basic form of the RUSLE calculation (Equation 1) (Wischmeier and Smith, 1978; Renard *et al.*, 1997) is defined as follows:

$$A = R \cdot K \cdot L \cdot S \cdot C \cdot P \qquad \textbf{(1)}$$

where,

A - potential soil erosion (computed annual average soil loss in t ha^{-1} $year^{-1}$);

R - rainfall and runoff erosivity factor (MJ mm $ha^{-1}h^{-1}$);

K - soil erodibility factor (t ha h ha^{-1} MJ^{-1} mm^{-1});

LS - slope length and gradient factor (dimensionless ratio);

C - vegetation cover factor (dimensionless ratio);

P - conservation/support practice factor (dimensionless ratio).

In this research, some adjustments in the RUSLE model were necessary to account for the effect of rainfall and vegetation seasonality. Because of this, and in order to investigate seasonal soil erosion, the rainfall-runoff erosivity (R) and vegetation cover (C) factors were analyzed on a seasonal basis, and the topographic (LS), support practice (P) and soil erodibility (K) factors were aggregated as ‘‘static’’. Soil erodibility (K factor) was considered invariant because soil properties are invariant or change very slowly throughout the year (Song *et al.*, 2005), especially in constant soil conditions (Bryan, 2000).

This section describes the RUSLE factors, provides details on the input information collected, and presents the equations to derive each factor. The methodology, which is summarized and structured in Figure 7, was used to investigate current scenarios of soil erosion. All the data necessary to obtain factors and soil erosion rates was processed with geographic information systems (GIS) (using ArcGIS software), in order to easily obtain the spatial distribution that is essential for decision making. For this study, ArcGIS version 10.1 was utilized.

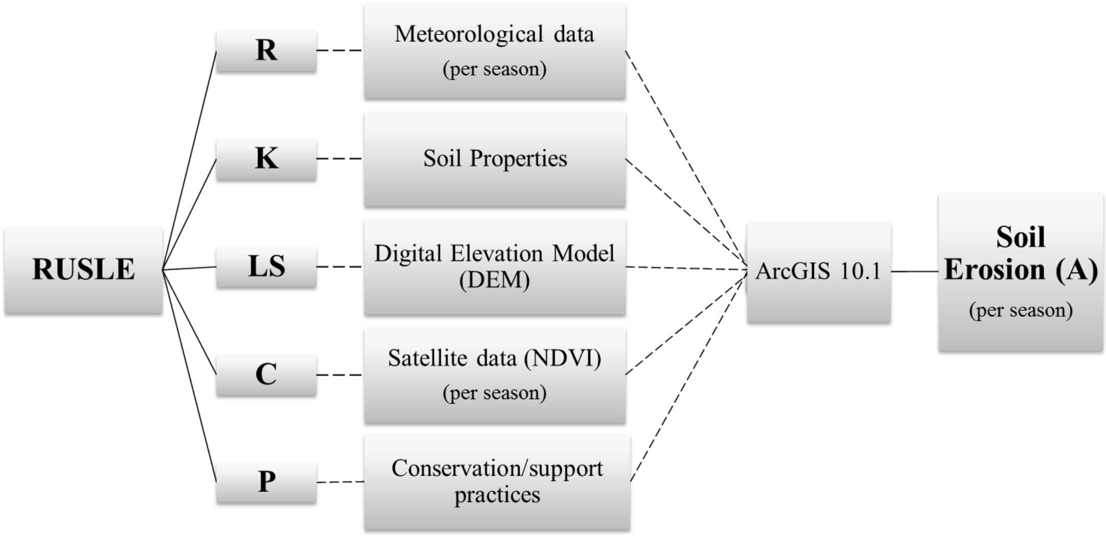

Figure 7 – Scheme of RUSLE's input factors to estimate soil erosion using ArcGIS.

2.2.1. Rainfall-Runoff Erosivity (R factor)

The rainfall–runoff erosivity (R factor) is typically recognized as one of the main indicators for the erosive potential of the impact of raindrops, also reproducing the potential of runoff generated by erosive rainstorms. In other words, it represents the energy and the intensity of rainfall as the driving force behind soil erosion.

According to Renard *et al.* (1997), the rainfall–runoff factor is estimated through the sum of erosive storm values EI_{30} occurring during a mean year (Equation 2):

$$R = \frac{1}{n}\sum_{j=1}^{n}\sum_{k=1}^{m}(EI_{30})_k \qquad \textbf{(2)}$$

where,

R - the average of the annual rainfall-runoff erosivity (MJ mm ha^{-1} h^{-1} yr^{-1});

E - the total storm kinetic energy in a single event k (MJ ha^{-1});

I_{30} - the maximum 30-minutes intensity in a single event (mm h^{-1});

n – the number of years of records;

m – the number of erosive events in a given year *j*.

The accurate computation of each storm EI_{30} demands for high resolution measurements on a small period time. Detailed information about rainfall for a period of 15 or 30 minutes at a time is not usually available for all locations, as only monthly or daily data or available for study. Some authors have found good correlations between EI_{30} and Modified

Fournier Index (Fmod) (Arnuldus, 1977; Coutinho *et al.*, 1994; Ferro *et al.*, 1999) or between EI_{30} and rainfall rates (Tomás, 1997; Loureiro, 2000; Goovaerts, 1999a). Some of these correlations use annual rainfall and disregard seasonal fluctuations.

Goovaerts (1999a) presents a calibrated regression with high correlation values (r^2 = 0.92). The regression (Equation 3) establishes the relationship between the monthly EI30 values with monthly rainfall for days where precipitation exceeds 10 mm ($rain_{10}$) and monthly number of days where precipitation exceeds 10 mm ($days_{10}$):

$$EI30_{month} = 6.56 \times rain_{10} - 75.09 \times days_{10} \quad \textbf{(3)}$$

This relation was calibrated from parameter measurements recorded at 17 tipping-bucket raingauges (time resolution = 1 min) in the south of Portugal, between October 1992 and March 1995. EI_{30} values were computed for each raingauge as per the instructions provided by the RUSLE handbook (Wischemeier and Smith, 1978). Monthly EI30 values were obtained by the sum of EI30 for each erosive storm that occurred during the month. The rainfall parameters $rain_{10}$ and $days_{10}$ were derived from the raingauge recorded data.

It is through this type of correlation that it's made possible to take into account seasonal fluctuations. In the study area, the monthly EI30 values were estimated using daily and monthly values of precipitation from 25 meteorological stations in the region. Daily precipitation values for a 30 year period (1979/80–2008/09) were used. A period of 20–25 years is mentioned by Wischmeier and Smith (1978) to compute the average R factor, in order to account for cyclical rainfall patterns. The data was processed over different seasons, considering autumn (October to December), winter (January to March), spring (April to June), and summer (July to September).

Using interpolation techniques (geostatistics), in this case the ordinary kriging (OK) described in sub-section 2.3.2, rainfall-runoff erosivity maps (annual and per season) were created for the study area, and the means for each experimental area were estimated.

2.2.2. Soil Erodibility (K factor)

Soil erodibility (K factor) reflects how susceptible a soil is to experience erosion. It is defined as the average rate of soil loss per unit of rainfall-runoff erosivity index from a cultivated continuous fallow plot, on a 9 % slope 22.1 m long (Renard *et al.*, 1997). The

K factor is a quantitative value, which is a function of soil properties, such as soil texture, content of organic matter, soil structure and permeability. According to USLE literature, it is experimentally determined using an algebraic approximation (Wischmeier and Smith, 1978) (Equation 4):

$$K = [2.1 \times 10^{-4}(12 - OM) \times M^{1.14} + 3.25(s - 2) + 2.5(p - 3)]/100 \qquad \textbf{(4)}$$

where OM is organic matter content, s is soil structure, and p is permeability class. M is the product of the primary particle size fractions (%MSilt) × (%MSilt + %MSand), where %MSilt is percent modified silt (0.002–0.1 mm), and %MSand is percent modified sand (0.1–2 mm). Modified silt is the amount of silt particles and very fine sand, considered the most susceptible particles to erosion because they can be easily removed by the raindrop splash and runoff water. K factor is expressed with U.S. units and division with the factor 7.59 will yield values expressed in SI units of $t\ ha\ h\ ha^{-1}\ MJ^{-1}\ mm^{-1}$.

In this study, the soil erodibility factor was estimated using Equation 4. In order to obtain necessary soil property values, soil samples of about 1 kg from depths between 0 to 20 cm were collected in different experimental areas (presented in the previous section), with different land uses. To estimate the permeability, the field-saturated hydraulic conductivity (HCsat) was measured *in situ* using a double-ring infiltrometer.

The total number of samples and HCsat measurements per land use, in each experimental field, are shown in Table 3. The sample locations *in situ* were recorded using Global Positioning System (GPS).

Table 3 - The number of soil samples collected and the number of HCsat field measurements per land use in each experimental area.

Experimental area		**“H. Gregos”**		**“H. Roncão”**	**“H. Pico”**
Land use	**Montado (20 ha)**	**Lucerne cult. (33.5 ha)**	**Olive Orch. (57.5 ha)**	**Montado (782 ha)**	**Vineyard (30 ha)**
Soil samples (nº)	25	27	52	82	25
Field HCsat measures (nº)	6	6	6	31	9

Firstly, the samples were air-dried; then, in laboratory, they were dried for about 6 hours at 40ºC using a ventilated oven. In order to remove rocks or gravel, the samples were then passed through a 2 mm sieve. The particle-size distribution was determined by the Bouyoucos hydrometer method (Bouyoucos, 1962), and OM, using the Walkley and

Black (1934) method, a wet oxidation procedure. The permeability class and soil structure class were defined in accordance with Renard *et al.* (1997).

The susceptibility of soil erosion and land degradation depends largely on various inherent soil properties, namely chemical, physical, biological and mineralogical properties (Cambardella *et al.*, 1994; Pérez-Rodríguez *et al.*, 2007), despite the limited soil variables used by the RUSLE model to characterize soil erodibility (K factor). To better understand the complex soil interactions and processes resulted from different land use and management practices, other soil chemical properties were analyzed, namely total nitrogen (N), soil pH and electrical conductivity (EC). The N content was determined according to Kjeldhal digestion, distillation and the titration method (Bremner and Mulvaney, 1982). Soil pH and EC were measured with glass electrode in a 1:2.5 soil/water suspension (Watson and Brown, 2011).

All soil data were introduced in the ArcGIS environment, and geostatistical analysis was performed using the Geostatistical Analyst Tool in order to obtain interpolation distribution maps (geostatistic methods are better explained in sub-section 2.3.2).

2.2.3. Slope Length and Steepness (LS factors)

The effect of the local topography on soil erosion is accounted for by the LS factor in the RUSLE methodology, which combines the effects of a slope length factor (L) and a slope steepness factor (S). The slope length is defined as the horizontal distance from the origin of overland flow to the point where either the slope gradient decreases to the point where deposition begins or runoff becomes concentrated in a channel. The slope steepness determines the influence of slope gradient on soil erosion (Wischmeier and Smith, 1978).

Direct measurements *in situ* of slope and slope length were initially proposed to evaluate these factors (Renard *et al.*, 1997). However, this method is only suitable for small plots and parcels because intensive field measurements are obviously not feasible on a regional scale. At the watershed scale, the use of a digital elevation model (DEM) in GIS for data input is a better approach (Nekhay *et al.*, 2009).

To estimate this factor, a DEM of 30 m resolution (available from ESRI (2009)) was used as input using ArcGIS software. Tools available on the GIS software allow users to define slope steepness and slope length raster covers by a number of different methods. The combined LS factor (without units) was computed for the entire study area by means of the Raster Calculator under ArcGIS Spatial Analyst extension using DEM, following Equation 5, as proposed by Moore and Burch (1986):

$$LS = (facc \times cell\ size/22.13)^p\ (\sin\alpha/0.0896)^q \tag{5}$$

where:
p and q - empirical exponents (p = 0.4 and q = 1.3) (Moore and Wilson, 1992);
$facc$ - flow accumulation;
cell size - size of DEM grid cell (30m for this study);
α - slope degree value.

Flow accumulation signifies the accumulated upslope contributing area for a given cell; cells with high accumulation values are usually stream or river channels. Before the extraction of flow accumulation from the DEM, flow direction had to be analyzed. Both steps were done by using the correspondent tools under Hydrology section found under Spatial Analyst Tool Function in ArcGIS 10.1. The slope angles was similarly extracted from the DEM using Surface Analysis under the Spatial Analyst function.

2.2.4. Soil Vegetation Cover (C factor)

According to RUSLE literature, the C factor reflects the effect of vegetation on soil erosion rate (Renard *et al.*, 1997). Vegetation cover protects the soil by dissipating the raindrop energy before it reaches the soil's surface. As such, soil erosion can be effectively limited with proper management of vegetation, plant residue, and tillage (Lee, 2004). This factor ranges between 0 and 1 (without units), and it is 1 for bare soil and approximates zero as surface cover increases. It bears a close linkage to land use types.

The C factor has been one of the most difficult RUSLE coefficients to estimate. According to RUSLE methodology, it is because it is based on soil loss ratios (SLR's), which are the ratios of soil loss at any given time in the cover management sequence to soil loss from the standard condition. SLR's are driven from the product of five sub-factors: prior land use, canopy cover, surface cover, surface roughness, and soil moisture (Renard *et al.*, 1997). To estimate these sub-factors, empirical equations can be used. However, the data necessary is usually not available or difficult to obtain (Bühlmann, 2006).

Several methodologies have been developed for estimating vegetation cover as an alternative to C values derived from RUSLE methodologies. Remote-sensing has been one of the most widely used methods for mapping the C factor (De Jong, 1994; Kouli *et al.*, 2009; Prasannakumar *et al.*, 2011). Vegetation cover can be estimated using vegetation indexes derived from satellite images, such as the Normalized Difference Vegetation Index (NDVI). The NDVI is an important indicator of vegetation growth and ranges from -1 to 1 (Van der Knijff *et al.*, 1999). This method gives a different perspective on soil erosion studies because it allows the estimation of intra-annual changes in vegetation cover through images for different periods (Ouyang *et al.*, 2010).

In this research, Landsat 5 Thematic Mapper (TM) and Landsat 7 Enhanced Thematic Mapper Plus (ETM+) imagery with 30 m resolution (USGS, 2012) was used to analyze vegetation cover in each land use. Table 4 presents the timeframe and the satellite data set acquired for each experimental area studied. Satellite images were taken each season during the aforementioned period due to the difficulty in retrieving data for each month. For each experimental site, a total of 12 satellite images (corresponding to a 3 year period) were used, in order to obtain the mean NDVI values per season.

Table 4 - Data set and acquisition period of satellite imagery used to obtain NDVI values, for each experimental study area.

Experimental area	Data set	Period of Acquisition
"Herdade do Roncão"	Landsat 7 ETM+	October 2006 to September of 2009
"Herdade dos Gregos"	Landsat 5 TM	January 2009 to December of 2011
"Herdade do Pico"	Landsat 5 TM	January 2009 to December of 2011

Landsat TM images were processed with IDRISI software (Eastman 2006), and NDVI values were calculated utilizing band 3 (red) and band 4 (near-infrared) as follows (Equation 6):

$$NDVI = (Band4 - Band3)/(Band4 + Band3) \quad \textbf{(6)}$$

Many methods have been developed to estimate the C factor using NDVI information, involving a linear or non-linear regression equation model (De Jong, 1994; Van der Knijff *et al.*, 1999). The regression equation is derived from the correlation analysis between the C factor values, measured in the field using the line-point intercept sampling method (Herrick *et al.*, 2005), and a satellite-derived NDVI (De Asis and Omasa 2007). In the present study, Equation 7 proposed by Van der Knijff *et al.* (1999) was used to generate a C factor surface from NDVI values:

$$C = e^{-\alpha\left(\frac{NDVI}{\beta - NDVI}\right)} \qquad \textbf{(7)}$$

where α and β are dimensionless parameters that determine the shape of the curve relating to NDVI and the C factor. Van der Knijff *et al.* (1999) found that this scaling approach provided better results than assuming a linear relationship, and the values of 2 and 1 were indicated for the parameters α and β, respectively. Using NDVI mpas, the C factor maps were produced for different seasons with Raster Calculator tool in the ArcGIS software.

2.2.5. Conservation/Support Practice (P factor)

The conservation/support practice factor (P) reflects the effects of specific practices that can be used to reduce the amount and rate of erosion. By definition the P factor is the ratio of soil loss with a specific support practice to corresponding loss with upslope and downslope tillage. As such, the lower the P factor is, the better the practice controls erosion.

The practices included are contouring (tillage and planting on the contour), strip-cropping, terracing, and subsurface drainage. These practices affect erosion by modifying the flow pattern, grade, or direction of surface runoff and by reducing the amount and rate of runoff (Renard *et al.*, 1997). P factor does not consider improved tillage practices such as no-tillage and other conservation tillage systems, sod-based crop rotation, fertility treatments and crop-residue management. Such erosion control practices are considered in the C factor.

For current conditions, in the different experimental land uses studied, no relevant support practice that allows to reduce soil erosion was identified. For these reason, the P factor is assigned a value of 1. However, when studying future soil erosion scenarios (sub-section 2.4.2), support practices were taken into account in order to construct a desirable scenario.

2.3. SPATIAL VARIABILITY OF SOIL PROPERTIES AND ERODIBILITY

Spatial variability is a well-known phenomenon of natural ecosystems as a result of complex interactions between geology, topography and climate. Moreover, the spatial variability of soil properties, which influences soil susceptibility to erosion, is highly related with anthropogenic factors, particularly in cultivated lands (Paz-González *et al.*, 2000; Wang and Shao, 2013). Information on the spatial variability and the interactions between soil properties, then, is essential for understanding the ecosystem processes and planning sustainable soil management alternatives for specific land uses (Pérez-Rodríguez *et al.*, 2007; Ziadat and Tamimeh, 2013).

Classical statistics and geostatistics methods have been widely applied on studies about spatial distribution of soil properties (Pérez-Rodríguez *et al.*, 2007, Tesfahunegn *et al.*, 2011). Adding to the methods in this study, HJ-Biplot was also used, which helped provide an added value for analyzing spatial variability.

2.3.1. Statistics

Variability was initially quantified using the conventional statistical approach, which assumes that the observations of a given property are statistically independent regardless of their spatial location, and the observations were used to obtain the property distribution.

Data was subjected to classical analysis using SPSS 17.0 software to obtain descriptive statistics, namely the mean, minimum (Min) and maximum (Max) values, standard deviation (SD), coefficient of variation (CV) and skewness of each property. Skewness is the most common statistic parameter to identify a normal distribution that is confirmed with skewness values varying from − 1 to + 1. When data showed a non-normal distribution, data transformation to normal distribution was applied before applying geostatistics to obtain distribution maps (through a specific tool available in the Geostatistical Analyst extension in ArcGIS software).

Duncan’s multiple-range test was applied to test the significant differences of soil property means between land uses, at the significance statistical level of 5%. When there is no significant differences among land uses, the same are combined within the same group.

2.3.2. Geostatistics

The aim of many researches is to estimate environmental variables at unsampled locations and map them. Geostatistics is a branch of applied statistics that focuses on the characterization of the spatial dependence structure of these environmental variables, when the number of samples is limited (Wackernagel, 1995; Webster and Oliver, 2001; Atkinson and Lloyd, 2010). The geostatistical methods use the stochastical theory of spatial correlation, both to predict at unsampled locations and to assess the uncertainty of these predictions (Goovaerts, 1999b; Burrough, 2001). It assumes that variables are correlated as a function of distance, (*i.e.* the sample values are not independent of each other, and one sample value gives some information about its neighbouring data point [Wackernagel, 1995]).

Geostatistical prediction techniques have been used in combination with GIS in soil science investigations due to their flexibility to obtain interpolation soil erosion maps (Diodato and Ceccarelli, 2004). The scientific information on the spatial variability and the interactions between soil properties is essential not only for soil erosion simulations, but also to allow for the understanding of the ecosystems' susceptibility and their processes, and planning sustainable soil management alternatives for specific lands (Pérez-Rodríguez *et al.*, 2007; Tesfahunegn *et al.*, 2011; Ziadat and Tamimeh, 2013). Geostatistical techniques have shown to be essential in obtaining degradation maps, since they allow for the interpolation from experimental points and estimating spatial errors and uncertainties (Diodato and Ceccarelli, 2004).

Soil data with the corresponding location was introduced in the ArcGIS environment and geostatistical analysis were performed using Geostatistical Analyst Tool, in other to examine spatial distribution of soil properties.

2.3.2.1.<u>Ordinary Kriging (OK)</u>

The Ordinary kriging method was selected for this study because it minimizes the error variance with unbiased estimates (Isaaks and Srivastava, 1989). Kriging is one of the most widely used interpolation geostatistical methods which assumes that variables close in space tend to be more similar than those further away. Kriging attempts to have a mean residual error equal to zero with the lowest possible value of the standard-deviation of the error, and at the same time it estimates the weighted linear combinations (w_i) of the available data ($Z(X_i)$) for the interpolation result ($Z(X_0)$) as it is shown in Equation 8 (Wackernagel 1995).

$$Z(X_0) = \sum_{i=1}^{n} w_i \, . Z(X_i) \;\wedge\; \sum_{i=1}^{n} w_i = 1 \quad \textbf{(8)}$$

The linear weight necessary for the interpolation is obtained by the ordinary kriging following the Equation 9 (Wackernagel 1995).

$$\begin{array}{ccccc} C & . & w & = & D \\ \begin{bmatrix} C_{11} & \cdots & C_{1n} & 1 \\ \vdots & \ddots & \vdots & \cdots \\ C_{n1} & \cdots & C_{nn} & 1 \\ 1 & \cdots & 1 & 0 \end{bmatrix} & \cdot & \begin{bmatrix} w_1 \\ \vdots \\ w_n \\ \mu \end{bmatrix} & = & \begin{bmatrix} C_{10} \\ \vdots \\ C_{n0} \\ 1 \end{bmatrix} \\ (n+1)*(n+1) & & (n+1)*1 & & (n+1)*1 \end{array} \quad \textbf{(9)}$$

In the above equation, matrix C contains the co-variances from all samples surrounding the sample to be interpolated. Matrix w contains the weights as well as a parameter called Lagrange Parameter. Matrix *D* contains the co-variance from the sample to be determined and the surrounding samples. The final objective of the ordinary kriging interpolation is to determine matrix w**.**

2.3.2.1.<u>Semivariogram</u>

The geostatistical methodology is based on the creation of a semivariogram (SV), which expresses the spatial dependence among samples (Chilés and Delfiner, 1999). The semivariogram is a plot between the distances of ordered data and their value of semivariance (Isaaks and Srivastava 1989). This plot explains the spatial relation between samples, and is exemplified by the following Equation 10 (Clark, 1979):

$$\gamma(h) = \frac{1}{2N_{(h)}} \sum [\, Z_i - Z_{i+h}]^2 \quad \textbf{(10)}$$

where γ(h) is the variance (the most related samples have lower values of variance), N(h) is the number of samples that can be grouped using vector h, Zi represents the value of the sample, and Zi+h is the value of another sample located at a distance h from the initial sample Zi (Chiles and Delfiner, 1999).

For a quantitative description of these features, it is useful to fit standard models (*e.g.* linear, shepherical, exponential) to the semivariance functions (Wackernagel, 1995) and the selection is usually performed by employing the cross-validation technique, which permits assessing the prediction‘s accuracy. The model fitted provided two important parameters that describe the spatial structure and dependence. Nugget is the variance at distance zero and reflects the sampling error. Sill is the semivariance value at which the semivariogram reaches the upper bound and flattens out after its initial increase; it is the variance in which the samples are no longer spatially related at the study area.

2.3.2.2.<u>Cross-validation</u>

Using the Geostatistical Analyst Tool (ArcGIS) and selecting the OK methods, a semivariogram was created for each measured property. The SV model selection was performed by employing the cross-validation technique, which permits the evaluation of the prediction accuracy. Cross-validation was executed to investigate the prediction performances through the statistical values, as the mean error (ME) (or root-mean-square standardized error [RMSSE]), which results from comparing the estimated semivariogram values and real observed values. Nugget and Sill were also analyzed to better understand the spatial dependence of each variable. If the ratio between Nugget and Sill is low (<0.25) then the samples are spatially correlated; if the ratio is high (>0.75) then the samples have a very low spatial correlation (Cambardella *et al.*, 1994).

Once cross-validation process was completed, interpolation maps of spatial distribution, for each soil variable, were produced according to the semivariogram model selected in the ArcGIS software.

2.3.2.3.<u>Trend analysis</u>

Sometimes, the effect of some factors is at least one order of magnitude greater (as topography or soil type) than the land use, causing directional trends. These trends affect all measurements in a deterministic way (nonrandom) where properties vary as a function of their coordinates (Tesfahunegn *et al.*, 2011). Trend in the variation signals a departure from the intrinsic hypothesis in which the process is assumed to be random and violates the assumptions on which geostatistics are based (McCormick *et al.*, 2009).

Thus, trend removal is necessary prior to the ordinary kriging method in order to create more accurate predictions, since the trend will not be influencing the spatial analysis. Trend analysis and removal of the soil properties was performed using the Geostatistical

Analyst of ArcGIS 10.1 as well. It provides a three-dimensional perspective of the data, and global trends are detected if a curve that is not flat (*i.e.*, a polynomial equation) can be fitted into the data. The deterministic component is subtracted from the data and residuals (the stochastic component) might be stationary or intrinsic (Chilés and Delfiner, 1999).

2.3.3. HJ-Biplot

This multivariate statistical technique allows for the graphical representation of a large data matrix (Gabriel, 1971), because it permits the interpretation of relations between individuals and between variables, as well as between both. Biplot is an exploratory data analysis, which can also indicate clustering of units with close characteristics, showing inter-unit distances as well as displaying variances and correlations amongst variables (Vallejo-Arboleda *et al.*, 2007; Gallego-Álvarez *et al.*, 2013).

Galindo-Villardón (1986) has developed HJ-Biplot, updating the Biplot methodologies introduced by Gabriel (1971). This is a symmetric representation technique similar to correspondence analysis, but not limited to frequency data. HJ-Biplot allows for a simultaneous analysis of variables and individuals (samples) in the same reference system, allowing not only the analysis of the behavior by sample, but also to determine which variable is responsible for such behavior (Galindo-Villardón, 1986; Gallego-Álvarez *et al.*, 2013). It allows for a graphical representation of dataset variability (samples and variables), without probabilistic distribution related assumptions.

The HJ-Biplot method has been used in many different research areas (García-Talegón *et al.*, 1999; González-Cabrera *et al.*, 2006; Martín-Rodríguez *et al.*, 2007; Gallego-Álvarez *et al.*, 2013; Gallego-Álvarez *et al.*, 2015). In the present study, this approach provides an additional value for analyzing spatial variability and establishing relations between soil properties and land uses.

A data matrix X suffers a factorization to reduce its dimensionality through single value decomposition, the algebraic base of biplot representation (Equation 11) (Gabriel, 1971).

$$X_{(n\times p)} = U_{(n\times r)}\,\Lambda_{(r\times r)}\,V'_{(r\times p)} \qquad \textbf{(11)}$$

where:

$\Lambda_{(r\times r)}$ is a diagonal ($\lambda_1, \lambda_2, \ldots, \lambda_r$) corresponding to the r eigenvalues of XX' or X'X;

$U_{(n\times r)}$ is an orthogonal matrix whose columns are the eigenvectors of XX';

$V'_{(r\times p)}$ is an orthogonal matrix whose columns are the eigenvectors of X'X.

With the *MultBiplot software*, developed by the University of Salamanca (Vicente Villardón, 2007), an HJ-Biplot was used to determine the relation between soil properties, land uses, and the correlations between both (soil properties and land uses), thereby defining patterns and clustering the samples in groups.

On the HJ-Biplot graphic representation, the points represent individuals (samples), and the vectors represent variables (in this case, chemical and physical soil properties). To interpret and discuss the graphs obtained with this methodology it's essential to be aware of (Gallego-Álvarez *et al.*, 2013):

- the distance between row points (samples) represents the variability and can be interpreted as similarity or dissimilarity, i.e. the close samples have similar behaviors;
- the angle formed by variable vectors (variables) is interpreted as correlation, i.e. small angles between variables represent similar behaviors with high positive correlations, and the obtuse angles that are almost a straight angle are associated with variables with high negative correlations (i.e., the cosine value of the angles characterizes the correlation between variables);
- the proximity of individual sample points and variable vectors means high preponderance; in other words, the closer a point is to a variable vector, the more important this sample is in explaining this variable;
- the length of the vector represents the variable's variability; the longer the vector, the higher the variability.

2.4. SOIL EROSION SCENARIOS

To support sustainable land use management and planning in the entire study area, the creation of future soil erosion scenarios is essential. The methodology for future scenario construction was divided in two different approaches well known as forecasting and backcasting. Both forecasting and backcasting procedures analyze future states; however, the process for generating this product is very different (Robinson, 1990). A forecast begins with the current situation and likely future paths, and then anticipates an end-state (Schwartz, 1996). The backcasting process firstly delineates a desirable future and then looks back to the present in order to identify strategies and plans to achieve it, examining the percentage at which adverse future can be avoided (Quist and Vergragt, 2006).

On this study, the forecasting scenarios of soil erosion were developed accounting the effect of climate change (affecting rainfall-runoff erosivity, R factor) and expected land use changes (affecting vegetation cover, C factor), on future erosion amounts estimated by RUSLE. Facing the expected increase of soil erosion from these changes, a backcasting scenario was constructed accounting the implementation of sustainable land management (SLM) practices (affecting conservation/support practice factor, P factor).

In order to obtain the different scenarios of soil erosion (current situation, forecasting and backcasting), each of the RUSLE factors were analyzed for the entire study area. The input RUSLE data was partially different to obtain the present scenarios than it was for experimental areas, considering not only a larger scale, but also the intention of studying future influences on R, C and P factors.

The C factor was analyzed using Corine Land Cover (CLC) data (IGEO, 2012), as it was easy to study future land use changes. The CLC provides information on the land cover using specific classes. The list of actual and expected future land use classes on the study area are presented on the Table 5, as the corresponding mean values for C factor.

Since the most recent CLC updated data is for 2006, this year was defined as the current situation. The forecasting and backcasting scenarios were studied for 2100, according to the available data for climate change.

The RUSLE factors, not accounting changes on R, C and P factors that are described in the next sub-sections, were computed as follows (current situation):

- R factor - the methodology presented in sub-section 2.2.1;

- K factor - high-resolution map (500 m) for Europe based on Land Use/Cover Area frame Survey (Panagos *et al.*, 2014) (values were calibrated with data obtained from experimental areas in the study area);
- LS factor - methodology as presented in sub-section 2.2.3;
- C factor – CLC classes were identified and the respective C factor values were assigned according Table 5;
- P factor - value of 1 (no support practice factor) for the entire study area, because the control practices are currently either nonexistent or insignificant in this area.

Table 5 - CLC classes and corresponding mean values of C factor (Wischmeier and Smith, 1978; Morgan, 2005).

CLC Class	Class Description	C factor
111	Continuous urban fabric	0
112	Discontinuous urban fabric	0
121	Industrial or commercial units	0
133	Construction sites	1
142	Sport and leisure facilities	0.05
211	Non-irrigated arable land	0.1
212	Permanently irrigated land	0.2
221	Vineyards	0.25
222	Fruit trees and berry plantations	0.25
223	Olive groves	0.25
231	Pastures	0.1
241	Annual crops associated with permanent crops	0.1
242	Complex cultivation patterns	0.1
243	Land occupied by agriculture, with significant areas of natural vegetation	0.1
244	Agro-forestry áreas	0.06
311	Broad-leaved forest	0.0015
312	Coniferous forest	0.0025
313	Mixed forest	0.002
321	Herbaceous natural grassland	0.01
321[a]	Sparse herbaceous vegetation and/or bare land	0.15
323	Sclerophyllous vegetation	0.003
323[a]	Xerophytic vegetation	0.1
324	Transitional woodland/shrub	0.003
324[a]	Xerophytic vegetation	0.1
512	Water bodies	0

2.4.1. Forecasting Scenarios

The forecasting scenario for soil erosion (2100) was constructed accounting the effects of climate change and land-use/cover change (LUCC). The next sub-sections describe the methodology used to analyze forecasted climate change and LUCC.

2.4.1.1. *Climate Change*

The climate change is likely to modify the global and regional patterns of precipitation and temperature. Climate change scenarios have been produced for decades at a global and regional scale, driven by global circulation models (GCMs) and regional climate models (RCM) (Santos *et al.*, 2002). In the south of Portugal (including the Alqueva area), according to the Hadley Centre Regional Model 2 (HadRM2), the total amount of precipitation is expected to decrease (by 11%) and temperatures are expected to increase (+5.9ºC) by the year 2100. Precipitation might be concentrated in the winter months, decreasing more noticeably during summer and autumn, and a 50 mm increase in the total number of precipitation days is expected by 2100 (Santos *et al.*, 2002).

Modifications in precipitation amounts and intensities are expected to affect the rainfall-runoff erosivity values (R factor of RUSLE) (Nearing *et al.*, 2004). Alpert *et al.* (2002) have debated an intensification in torrential rainfall despite a decrease in predicted annual rainfall in the western Mediterranean region. For the south of Portugal, in 2100, it is likely that an increase between 0 and 50% in annual runoff, associated to strong rainfall intensity resulting from the concentration of precipitation in a small number of events, will occur (according HadRM2) (Santos *et al.*, 2002). Considering these values, a mean increase of 12.5% on rainfall-runoff erosivity for 2100 was deliberated when developing the future (forecasting and backcasting) scenarios of soil erosion for the study area.

As a consequence of changes in precipitation patterns and temperature, the natural vegetation is equally expected to alter in the region, due to the vulnerability of some species to extreme conditions. According to the future scenarios of vegetation distribution in Portugal, developed according to ecosystem models, an increase of xerophytic and sparse vegetation in the study area is expected (Santos *et al.*, 2002). New classes were added while investigating the forecasting scenarios of LUCC, accounting for the influence of climate changes on existing land use classes (CLC). The new classes (Table 5) resulted from the transition of: 321-Natural grassland to 321a- Sparse herbaceous vegetation and/or bare land; 323-Sclerophyllous vegetation and 324-Transitional woodland/shrub to 323a and 324a-Xerophytic vegetation. The increase of these classes was considered for areas with the most arid conditions, namely low annual precipitation values (between 400 and 500 mm) and high temperatures (greater than 17.5%) (Samora-Arvela, 2013).

2.4.1.2.Land-use/cover change (LUCC)

Over the last few decades, some models were developed to study the forecasting LUCC (Veldkamp and Lambin, 2001; Parker *et al.*, 2003; Verburg *et al.*, 2004; Koomen *et al.*, 2007), exploring the implications of possible upcoming situations or directions (such as economic growth or ecological policy changes) on future land use (Verburg *et al*., 2004; Yu *et al.*, 2011). The multi-agent system models of land-use/cover change (MAS/LUCC models) have been widely used. This multiple methodology usually combines cellular automata models (CA) that represent landscape, with agent-based models (ABMs) to express human interaction on said landscape (Parker *et al.*, 2003).

The CA models simulate the spatial and temporal dynamics of future land use change based on observed modifications from the past. Markov chains are used to construct transition matrices (quantitative analysis of LUCC areas) and to determine trend dynamics that are projected in order to simulate future LUCC (Nainggolan *et al.*, 2012; Kamusoko *et al.*, 2009). This methodology considers a cellular entity that changes between states (land use types) according to the neighborhood conditions, and adopts transition rules in a pre-defined time period (Verburg *et al.*, 2004; Myint and Wang, 2006).

ABMs were used in order to incorporate the human factor in the LUCC simulations (Nainggolan *et al.*, 2012; Mena *et al.*, 2011). ABMs allow for the modelling of interactions between human and natural systems by specifying various decision-making agents (Valbuena *et al.*, 2010). These multiple agents, with different strategies and characteristics, interact both amongst themselves and with their ecosystem as well (Matthews *et al.*, 2007; Parker *et al.*, 2003).

The multi-criteria evaluation (MCE) is an agent-based technique to delineate agent's decision behaviors. This technique analyzes a relative static set of criteria (landscape factors and constraints) (Ligtenberg *et al.*, 2004). Thus, MCE is a method that analyses the suitability for potential transition of a certain area to maintain or change its land use given the human intentions or objectives, unfolding the influence and importance of each criteria.

The Analytical Hierarchy Process (AHP) is one of the most widely used methods to achieve criteria weights among different criteria based on decisions and objectives for a certain area (Saaty, 2004; Chen *et al.*, 2010). The combination of these methodologies with the Geographic Information Systems (GIS) technology is a suitable tool because of its ability to integrate a great amount of heterogeneous data at multi-spatial and multi-temporal scales (Parker *et al.*, 2003).

The scenarios of future LUCC in the Alqueva area were performed using MAS, more specifically according the following steps (Samora-Arvela *et al.*, 2012):

1. Construction of transition matrices by a cellular automata model (CA). These Markov matrices were produced with IDRISI software, allowing for the analysis of past land use changes. The Corine Land Cover (CLC) data from the years 2000 and 2006 was used as input data in the process, considering the reservoir filling began in 2002, and only the changes between these years can reflect the influence of dam construction. The calculated probabilities were used to define changes in individual land uses and also to build future scenarios.

2. Elaboration of suitability maps using MCE and AHP, resorting to digitized landscape induction factors and constraints in GIS, and weighing their influence in defining the most suitable areas for land use transition. These maps allow for the description of the decision making process according to four potential directions: biomass production for bioenergy, agriculture intensification, rural and golf tourism development, and the impact of climate change in vegetation cover. The studied factors were the planning intentions of Governmental Landscape Planning (PROZEA - Regional Plan for the Surroundings of Alqueva reservoir; PROTA - Regional Plan of the Alentejo Region; POAAP - the Plan for the Alqueva and Pedrogão reservoirs), topography, villages and historic components, proximity of accessibilities and constraints such as protected areas. Conjugation of criteria and the weights associated determine the areas which are more suitable to alteration.

3. Combination of transition probabilities (2000-2006) and agent-based suitable maps in order to assign LUCC in future landscape related scenarios (forecasting).

The LUCC scenarios were obtained for 2050 and 2100, and enable the reflection regarding the implications of these changes on soil erosion by the variation of RUSLE's C factor (referring to vegetation cover). The LUCC scenario for 2100 was accounted on the forecasting and backcasting scenario of soil erosion for the same year.

2.4.2. Backcasting Scenario - Sustainable Land Management

The forecasting research improved the understanding of soil erosion vulnerability at the Alqueva watershed and the influence of different factors. The aim is to use this knowledge as a basis to promote sustainable land management (SLM) and land use planning in the region.

With that in mind, a backcasting scenario was created considering the application of some SLM practices, particularly on the land uses that were forecasted to increase in the region (namely irrigated land, vineyards, fruit trees and olive groves, and sparse and xerophytic vegetation). This backcasting scenario breaks expectations of future erosion vulnerability predictions and, along with backward inductions to the present, defines a desirable future with regard to soil erosion and sedimentation for the reservoir. Table 6 presents some common sustainable land use practices considered for each land use and their respective P factor value according to Wischmeier and Smith (1978) and Morgan (2005).

Table 6 - P factor considered for soil erosion scenario in 2100 considering the application of some SLM (Wischmeier and Smith, 1978; Morgan, 2005).

CLC Class	Class Description	Conservation/support practice	P Factor
212	Irrigated land	Contouring	0.5
221, 222 and 223	Vineyards, fruit trees, olive groves	Contouring + strip-cropping	0.25
323a and 324a	Xerophytic vegetation	Afforestation (more 50% cover)	0.5

Contouring can effectively reduce soil erosion in the areas with some slopes, since planting and tillage practices occur along the contours instead of in straight lines and downhill (Renard *et al.*, 1997). This practice decreases runoff, rill formation, and increases water infiltration (Liu *et al.*, 2011).

Using strip cropping, row crops (or trees) and protection-effective crops are grown in alternating strips aligned with the contour. This technique requires specific plants (*e.g.* grass, special shrubs) between productive crops as a means of creating a denser ground cover which in turn hinders runoff. The use of natural vegetative strips or green covers has shown to be fundamental in reducing the soil erosion rates on vineyards and olive/other fruits trees (WOCAT, 2007; Schwilch *et al.*, 2012b).

Afforestation is aimed at stopping erosion and makes better use of rainfall in order to maintain the sustainability of agricultural systems, since it provides for better vegetation cover protection (Schwilch *et al.*, 2012b).

2.5. SEDIMENT DELIVERY

The RUSLE equation estimates only local erosion amounts and cannot be used to estimate the sediment yield for an entire study area. Since there is deposition between the source and a fluvial system (*e.g*. a reservoir), the rate of sediment yield carried by natural streams is much less than the gross erosion on its upstream watershed (Julien, 2010).

Significant research has been performed to estimate the annual sediment yield or the percentage of eroded sediment that is being effectively transported into rivers or reservoirs. At a regional scale the most widely used and accepted method is the sediment delivery ratio (SDR). The SDR is defined as the ratio of the sediment yield at a given stream cross section to the gross erosion from the watershed upstream from the measuring point (Julien, 2010). A generalized SDR curve (Equation 12) was developed from the data for 300 watersheds throughout the world (Vanoni, 1975). This curve is based on SDR-area power function equation.

$$SDR = 0.4724 \times A^{-0.125} \qquad \textbf{(12)}$$

where A is the area of the basin (km^2).

In order to measure the amount of sediments in the study area, which is going effectively into the Alqueva reservoir, the SDR ratio was estimated through the presented equation and multiplied to the average soil erosion rate. The sediment yield is obtained accounting soil erosion rate in different investigated scenarios: current situation (2006), forecasting scenario accounting land use and climate changes (2100), and backcasting scenario accounting sustainable land management (2100).

2.6. SIMULATION MODEL AND DECISION SUPPORT SYSTEM

RUSLE was used in this research despite some difficulties and limitations. The RUSLE model is usually applied to estimate annual soil erosion; however, in the present study, season variations were estimated facing the high evidence of their existence in

Mediterranean regions. Additionally, despite the complexity of erosion processes, RUSLE considers the interdependence of each factor and does not take into account all conservation practices (Cakula *et al.*, 2012). Mostly due to these limitations, an alternative simulation model to estimate soil erosion was developed in this investigation (Cakula, 2012).

The alternative simulation model was created with close connection to a decision support system (DSS) in order to improve its usage. It enables erosion risk assessment under various situations and it is important for local governments to be able to present land owners with the consequences of their actions and decisions. The system intend to improve their decision making process.

The conception of dynamic system consists of two parts:

- the creation of a simulation model in the Stella 9.1.3 modeling environment for soil erosion assessment (as an alternative to RUSLE);
- the development of an associated decision support system with PHP and AJAX programming to provide recommendations based on simulation results.

Both parts are connected through Microsoft Office Excel spreadsheets. The Stella 9.1.3 modelling environment is used because this software is easy to manage, is object-oriented, and works successfully with various functions and calculations (Cakula, 2012).

2.6.1. Simulation Model Structure (Stella)

Facing the complexity of soil erosion, several interconnected sub-systems should be considered in order to simulate soil erosion. After a thorough analysis of soil erosion processes and local condition presented on Cakula (2012), seven sub-systems were defined, and their respective descriptions are presented summarily in Table 7.

The processes, equations and parameters for each sub-system are clarified by Cakula (2012) and published in Panagopoulos *et al.* (2014). All these processes were modelled separately with Stella 9.1.3 software; however, they are interconnected, and the user needs only to run this simulation once (managed by a graphical user interface (GUI)). The simulation model's complete scheme is presented by Cakula (2012).

Table 7 - Systems requiring modelling in order to predict erosion.

Modelled system	Description
Natural vegetation cover	Models variances in natural vegetation during the year.
Agricultural vegetation cover	Six different farming methods are examined.
Soil infiltration capacity	Models how much water soil can absorb at any given point. Includes OM model.
Surface water	Water that cannot be absorbed by soil becomes runoff.
Runoff sedimentation	Models the amount of sediments caused by water moving on the surface of the soil.
Raindrop sedimentation	Models the amount of sediments from rain-splash.
Soil amount	Soil amount and sedimentation per ha.

2.6.2.Input Scenarios and Decision Support System

In order to improve the use of the simulation model that was created, a decision support system (DSS) was associated to provide land management recommendations according to the simulated soil erosion results. A graphic user interface (GUI) was created as a component of a DSS, thus developing a link between the decision maker and the model. A web-based interface was used because it is easily accessible to all decision makers who should require it (without installation). It is also a familiar environment for most people, which can reduce any perceived complexities.

Using this DSS tool is fairly similar with using the actual model; however, access to the model itself is not provided when using it. This reduces the possibility of an accidental modification of the model by users. The specific information about servers and programing steps for can be accessed consulting Cakula (2012).

The model that the created dynamic system will operate has been defined as having four major groups of necessary parameters: soil characteristic data (texture, structure, organic matter, hydraulic conductivity), meteorological data, topography data, and land use practice data. In order to assist decision makers save time in the “Herdade do Roncão” experimental site, where a tourism project is being implemented, specific scenarios considering 24 sampling sites have been upload to the dynamic system. The data used for these “Herdade do Roncão” scenarios was as follows:

- specific soil characteristics (texture, organic matter, hydraulic conductivity) obtained from collected samples;

- topography data obtained with ArcGIS Spatial Analyst tools, to find slope degree values and flow direction for an area of 1 square ha;
- meteorological data for the region, obtained from 25 meteorological stations in the Alqueva reservoir area for a 30 year period;
- Natural vegetation as land use for these scenarios.

However, one of the objectives of the dynamic system is also to allow data to be modified and saved in the GUI, in order to be used by decision makers in other lands in this Alqueva area.

3. RESULTS AND DISCUSSION

3.1. SOIL EROSION BY RUSLE - DIFFERENT LAND USES

3.1.1. Rainfall-Runoff Erosivity (R factor)

Rainfall-runoff erosivity values were estimated for 25 stations in the surrounding area of the Alqueva using daily precipitation values throughout a 30 year period (1979/80–2008/09) (SNIRH, 2011). The monthly $rain_{10}$ (rainfall for days where precipitation exceeds 10 mm) and monthly $days_{10}$ (number of days where precipitation exceeds 10 mm) were estimated, and the R factor was obtained using these parameters through the Equation 3. Table 8 presents the estimated mean values of R factor during these 30 years (monthly, seasonal and annual means) for each meteorological station in the region.

Using the R factor values estimated for 25 meteorological stations, geostatistic maps (a continuous surface) were obtained for the entire study area (surrounding reservoir area), using the ordinary kriging (OK) interpolation method (see sub-section 2.3.2). Figure 8 displays these R factor interpolation maps (annual and seasonal maps) obtained with the use of geostatistic tools. It is noticeable that value R factor in the study area was characterized by spatial variability with areas with greater values (particularly in the north and western part) than others (eastern part). Comparing with the topography map (sub-section 2.1.2, Figure 3), it is noticeable that areas with high R factor values are the areas with greater altitude, which can increase the risk of soil erosion in general.

The study area is essentially affected by a strong seasonality effect. The highest R factor values were associated with the first rainfall events during autumn (467.8 MJ mm ha^{-1} h^{-1}), and the lowest values occur during summer (77.8 MJ mm ha^{-1} h^{-1}). In winter, R factor values decrease slightly when compared with autumn (211.4 MJ mm ha^{-1} h^{-1}), and continue decreasing during the spring months (152.6 MJ mm ha^{-1} h^{-1}). Annual rainfall-runoff erosivity in the study area was found to be 912.6 MJ mm ha^{-1} h^{-1}. Percentage wise, about 51% of annual R factor occurs in autumn, 23% in winter, 17% in spring, and only 9% during summer. These values and seasonal trends are comparable to the results found in other studies in Mediterranean regions (Van der Knijff *et al.*, 1999; Diodato, 2005) and in the Iberian Peninsula (Angulo-Martínez and Beguería, 2009; Beguería *et al.*, 2009).

Table 8 - Estimated rainfall-runoff erosivity values (MJ mm ha^{-1} h^{-1}) (monthly, seasonal and annual) for 25 meteorological stations (for the period between 1979/80 and 2009/10).

			Autumn			Winter			Spring			Summer			Annual Mean (MJ mm ha^{-1} h^{-1})
	X	Y	October	November	December	January	February	March	April	May	June	July	August	September	
Alandroal	638800	4283918	174.1	191.7	170.5	104.1	67.0	52.8	66.9	61.5	25.6	5.6	4.8	77.4	1001.9
Amareleja	655016	4230613	142.5	157.9	103.4	66.8	44.8	19.5	58.0	40.8	28.4	1.3	4.4	85.3	753.2
Azaruja	606547	4284492	173.6	155.6	114.2	92.2	50.4	52.0	41.4	44.1	33.9	0.9	17.3	50.1	825.8
Barrancos	675171	4222358	103.4	148.3	123.4	112.5	65.5	40.7	101.5	48.9	40.7	2.5	11.3	90.9	889.5
Caia	665675	4306009	138.2	89.5	100.2	65.5	47.6	59.5	40.3	53.0	43.5	0.6	1.5	54.4	694.0
Cuba	597450	4225037	171.5	180.9	126.2	117.0	79.1	51.9	66.9	46.2	13.0	7.0	15.6	61.3	936.5
Ferreira (Capelins)	642687	4269979	143.1	161.7	130.0	104.1	56.1	50.0	69.0	57.5	26.4	4.5	6.2	51.5	859.9
Herdade de Valada	637729	4201213	137.1	162.6	119.0	65.6	47.8	40.6	61.8	63.0	11.4	2.6	10.8	99.0	821.2
Juromenha	652791	4289281	212.8	175.4	130.3	85.2	49.6	47.2	40.2	58.1	11.8	2.6	6.6	58.4	878.1
Montoito	622321	4263558	153.6	160.4	121.3	104.4	60.2	53.0	92.3	73.4	44.4	6.6	7.6	59.2	936.3
Pedrogão	618603	4219661	140.0	160.2	104.8	91.6	63.7	44.6	63.3	36.7	28.2	4.3	6.2	64.7	808.2
Portel	612963	4240557	181.7	174.2	119.9	93.7	87.2	47.5	85.8	52.1	11.7	3.8	6.1	74.6	938.2
Reguengos	628591	4253799	149.6	131.5	106.1	95.8	50.9	53.5	62.4	44.4	19.6	3.4	9.2	64.5	790.9
Rosário	643885	4276044	121.7	212.8	158.1	107.9	42.5	38.1	50.7	59.2	18.1	2.9	1.6	57.9	871.6
Salvada	606518	4198408	137.9	204.8	164.5	106.6	65.5	72.4	89.1	62.0	18.4	0.0	1.4	69.7	992.4
Sta. Clara Louredo	598873	4202668	147.9	140.2	113.2	78.2	62.4	48.3	83.1	36.5	14.9	0.3	0.2	64.6	789.9
Santa Iria	626898	4193704	136.7	224.1	108.2	82.3	46.9	31.7	45.1	47.8	12.9	6.9	2.4	53.6	798.4
Santa Susana	616901	4270469	181.5	185.5	122.9	103.7	56.9	27.3	71.4	65.8	54.6	3.6	12.2	59.6	944.9
Santiago Maior	632204	4267085	210.7	166.3	143.7	124.9	67.2	61.2	81.1	68.9	29.8	3.3	10.6	83.6	1051.4
Sto. Aleixo Restaur.	662372	4214414	122.8	137.4	113.5	88.1	45.8	42.4	77.4	61.1	56.6	2.9	9.9	59.4	817.3
Serpa	622678	4200370	178.5	239.4	115.8	107.5	34.4	59.1	84.9	44.9	12.4	1.5	1.8	69.8	950.0
Sobral da Adiça	652736	4209241	147.7	209.5	143.0	95.9	76.3	44.5	100.7	63.5	34.5	3.2	4.8	68.3	992.0
São Manços	608972	4257594	165.7	169.2	129.6	97.7	64.6	38.1	61.8	44.4	18.3	0.8	12.0	66.1	868.3
Vidigueira	604778	4229456	194.9	213.5	198.7	179.0	124.3	68.3	96.5	59.1	25.4	17.0	9.6	79.2	1265.5
Vila Viçosa	637187	4294120	231.1	253.4	211.5	168.8	76.9	66.8	92.2	84.0	20.2	4.6	4.7	56.5	1270.7
Monthly mean (MJ mm ha^{-1} h^{-1})			159.9	176.2	131.7	101.6	61.3	48.4	71.4	55.1	26.2	3.7	7.1	67.2	**912.16**
Seasonal Mean (MJ mm ha^{-1} h^{-1})			**467.8**			**211.4**			**152.6**			**77.8**			

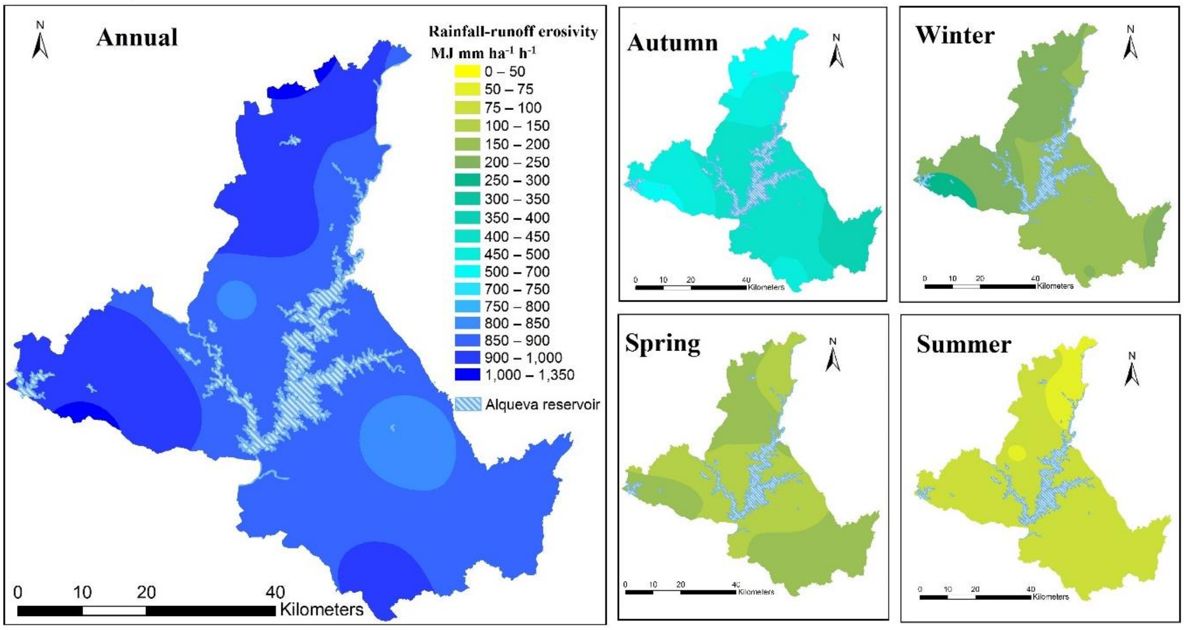

Figure 8 – Annual and seasonal distribution maps of rainfall-runoff erosivity for all the study area.

The maps created for all the surrounding area of Alqueva reservoir were used when predicting soil loss for each experimental site studied. The mean values of R factor for each experimental site are presented in Table 9. As one can see, values vary slightly between areas despite their proximity. The highest annual mean value of R factor belonged to "Herdade dos Gregos" (892.0 MJ mm ha^{-1} h^{-1}), followed by "Herdade do Roncão" (875.6 MJ mm ha^{-1} h^{-1}) (both in the western side of the reservoir) and the lowest for the "Herdade do Pico" (851.7 MJ mm ha^{-1} h^{-1}) (eastern side).

Table 9 - Mean values of R factor (seasonal and annual) for all experimental study areas (MJ mm ha^{-1} h^{-1}).

Season	"Herdade dos Gregos"	"Herdade do Roncão"	"Herdade do Pico"
Autumn	445.7	423.8	404.1
Winter	212.2	193.4	170.0
Spring	152.6	146.4	158.7
Summer	79.9	79.7	81.2
Total (MJ mm ha^{-1} h^{-1})	**892.0**	**875.6**	**851.7**

3.1.2. Soil Erodibility (K factor) and Spatial Variability of Soil Properties

The samples collected in experimental sites for different land uses were analyzed in laboratory to estimate several soil properties and estimate soil erodibility values. The analysis is shown together for all experimental areas with different land uses to facilitate comparisons and debate.

Along with the soil erodibility estimation, the statistics, spatial distribution and correlations of some chemical and physical soil properties were analyzed to better understand the effect of different land use and management practices on its spatial variability.

3.1.2.1.Statistics parameters analysis

The descriptive statistics (mean, coefficient of variation [CV], minimum [Min], maximum [Max] and Skweness) and geostatistical parameters (semivariogram [SV] model, nugget, sill, mean error [ME] and the root-mean-square standardized error [RMSSE]) of measured soil properties and estimated K factor, for each land use, are presented in Table 10. Some soil properties were only measured for determined land uses as shown. Duncan's test showed significant differences between land use means of all analyzed properties. Mean values with different letters in the same property are significantly different between land uses at $p \leq 0.05$. This reveals that land uses might have highly significant effects on soil properties and, in turn, on soil erodibility.

All of the statistic parameters help better understand the variability within each land use and correlations with the specific land use practices. All measured properties varied considerably within the areas as indicated by the CV (varying between 4.2 and 116.1%). Nitrogen (N), organic matter (OM), field-saturated hydraulic conductivity (HCsat) and very fine sand (VFS) show the highest variation values, especially for cultivated fields (lucerne cultivation, olive tree orchard and vineyard), which can be explained by the lack of homogeneous fertilization or tillage practices applied to soil in these sites. The skewness results, which vary from -1.48 to 3.54 in this study, indicated that some soil properties of different uses were not normally distributed, especially OM and N. The main reason for some soil properties having abnormal distributions may be related to soil management practices (Tesfahunegn *et al.*, 2011). As mentioned, data was transformed to normal distribution when necessary.

Table 10 – Conventional statistics and geostatistic parameters for soil properties and erodibility (K factor).

			CONVENTIONAL STATISTICS					GEOSTATISTICS					
			Mean	**CV (%)**	**Min**	**Max**	**Skew-ness**	**SV model**	**Nugget**	**Sill**	**Nugget/ Sill**	**ME**	**RMSSE**
"HERDADE DOS GREGOS"	Montado grassland	**Clay** (%)	17.29^{C}	37.7	5.68	29.62	0.07	Exponential	0.00	38.30	0.00	0.0056	1.01
		Silt (%)	29.55^{B}	17.2	12.99	39.72	-0.99	Exponential	0.00	36.00	0.00	0.0238	1.05
		Sand (%)	53.16^{A}	13.5	39.68	70.34	0.33	Pentaspherical	0.00	57.60	0.00	0.0223	1.00
		VFS (%)	11.13^{A}	25.6	4.49	19.04	0.16	Stable	0.00	12.00	0.00	-0.0188	0.99
		OM (%)	5.22^{C}	32.1	2.25	10.35	1.19	Exponential*	0.03	0.07	0.44	-0.0003	1.05
		N (%)	0.19^{C}	43.2	0.07	0.42	1.13	Exponential*	0.06	0.18	0.32	0.0008	1.05
		EC (dS/m)	0.100^{A}	38.1	55.5	217.5	1.28	Exponential*	0.01	0.13	0.09	0.5640	0.95
		pH	5.90^{B}	4.2	5.38	6.30	0.01	Exponential	0.00	0.07	0.00	0.0022	0.99
		HCsat (cm/h)	4.56^{A}	42.9	1.20	7.20	-0.57	-	-	-	-	-	-
		K (t ha h ha^{-1} MJ^{-1} mm^{-1})	0.021^{A}	31.4	0.006	0.039	0.43	Stable	0.00	0.00	0.00	0.0000	1.00
	Lucerne cultivation	**Clay** (%)	13.29^{B}	28.8	5.65	22.28	0.32	Stable	0.00	15.30	0.00	0.0018	1.02
		Silt (%)	33.79^{C}	26.6	8.35	47.29	-1.48	Stable*	0.00	44.20	0.00	0.0073	0.97
		Sand (%)	52.93^{A}	17.7	39.32	79.99	1.00	Exponential	0.00	92.00	0.00	0.0297	0.99
		VFS (%)	15.28^{B}	37.0	2.59	25.17	-0.39	Exponential	15.60	25.00	0.62	0.0347	1.04
		OM (%)	2.08^{B}	52.8	0.45	5.44	1.21	Exponential*	15.90	1.19	0.13	0.0036	0.95
		N (%)	0.11^{B}	70.2	0.02	0.35	1.43	Circular*	0.11	0.52	0.20	0.0018	1.02
		EC (dS/m)	0.107^{A}	45.9	40.5	205.0	0.64	Exponential	1.15	1.79	0.64	0.2240	0.96
		pH	7.14^{C}	4.3	6.53	7.85	0.02	Exponential	0.04	0.08	0.57	0.0052	1.07
		HCsat (cm/h)	5.95^{A}	26.7	0.65	1.30	-0.29	-	-	-	-	-	-
		K (t ha h ha^{-1} MJ^{-1} mm^{-1})	0.039^{C}	21.9	0.013	0.052	-0.88	Stable	00.00	0.00	0.00	0.0000	1.03
	Olive tree orchard	**Clay** (%)	9.83^{A}	28.8	5.40	16.66	0.52	Stable	0.00	8.04	0.00	0.0000	0.99
		Silt (%)	24.37^{A}	46.8	3.82	43.36	-0.41	Pentaspherical	50.00	89.80	0.55	0.0006	0.90
		Sand (%)	65.81^{B}	18.2	40.6	89.66	0.21	Exponential	0.00	161.00	0.00	0.0020	0.91
		VFS (%)	18.14^{C}	32.5	4.49	19.04	0.16	Exponential	0.02	33.70	0.00	0.0037	1.05
		OM (%)	2.10^{B}	52.8	0.62	8.35	3.54	Exponential*	0.07	0.17	0.44	-0.0007	1.02
		N (%)	0.10^{B}	45.3	0.04	0.29	2.02	Exponential*	0.02	0.15	0.12	0.0028	1.10
		EC (dS/m)	0.182^{B}	61.3	53.50	583.50	1.80	Exponential	0.00	1.40	0.00	0.6820	1.03
		pH	5.48^{A}	7.6	4.30	6.21	-0.43	Exponential	0.00	0.22	0.00	-0.0002	0.95
		HCsat (cm/h)	2.60^{A}	64.9	0.00	0.67	-0.45	-	-	-	-	-	-
		K (t ha h ha^{-1} MJ^{-1} mm^{-1})	0.038^{C}	33.6	0.012	0.061	-0.36	Exponential	0.00	0.00	0.51	0.0000	0.92
"HERDADE DO RONCAO"	Montado	**Clay** (%)	15.28^{B}	23.0	8.56	27.28	0.62	Pentaspherical	4.74	8.95	0.53	-0.0148	0.988
		Silt (%)	22.01^{A}	35.3	5.64	22.01	1.08	Exponential	42.61	15.56	2.74	-0.0362	1.099
		Sand (%)	62.71^{A}	13.4	36.47	80.80	-0.80	Pentaspherical	44.57	17.53	2.54	0.0754	1.093
		VFS (%)	13.52^{B}	53.3	0.00	31.77	0.40	Exponential	47.51	0.00	0.00	0.1246	1.096
		OM (%)	3.63^{C}	38.4	0.80	7.73	0.63	Exponential	0.89	1.11	0.80	-0.0007	0.989
		HCsat (cm/h)	15.12^{A}	116.1	1.50	62.40	1.51	Exponential*	0.00	10.38	0.00	-0.0028	0.983
		K (t ha h ha^{-1} MJ^{-1} mm^{-1})	0.023^{A}	37.8	0.003	0.047	0.55	Exponential	0.00	0.00	0.00	0.0000	0.989
"HERDADE D PICO"	Vineyard	**Clay** (%)	21.6^{D}	7.9	7.40	30.96	-0.39	Exponential	29.81	16.76	1.77	-0.1699	0.97
		Silt (%)	21.8^{A}	19.9	14.00	29.00	-0.02	Exponential	6.99	17.50	0.40	0.1434	1.05
		Sand (%)	56.5^{A}	29.9	49.40	65.60	0.36	Exponential	19.88	0.00	0.00	-0.0024	1.00
		VFS (%)	11.7^{A}	66.8	3.01	31.50	1.11	Pentaspherical	31.49	40.49	0.78	0.2542	0.92
		OM (%)	0.77^{A}	47.6	0.30	1.66	0.44	Exponential	0.00	0.00	0.00	0.0042	0.92
		N (%)	0.07^{A}	28.3	0.01	0.11	-0.78	Exponential	0.00	0.00	0.00	0.0000	1.03
		HCsat (cm/h)	4.21^{A}	47.2	1.20	7.20	-0.32	-	-	-	-	-	-
		K (t ha h ha^{-1} MJ^{-1} mm^{-1})	0.029^{B}	44.7	0.011	0.062	0.85	Exponential	0.00	0.00	0.00	0.0004	0.94

*Transformation for normal distribution.

Means with different letters (A, B, C and D) in the same property are significantly different between land uses at $p \leq 0.05$ (Duncan's test). CV - coefficient variation; Min - minimum; Max - maximum; VFS - very fine sand; OM - organic matter; N - Nitrogen; EC - Electrical conductivity; HCsat - field-saturated hydraulic conductivity; K - soil erodibility; SV – semivariogram; ME - mean error; RMSSE - root-mean-square standardized error.

Despite some significant differences, the soil's particle size distribution consists of all sandy loam formed mainly with sand (52-66%), followed by silt (21-34%) and clay (9-22%). The differences between land use areas can be explained not only by different soil types/profiles or topography, but also with the respective land use and management practices. The intensive cultivation (intensive irrigation, tillage and continuous cultivation, fertilizers and lime application) create conditions that promote changes in soil weathering and moisture, and consequently on soil texture (Yimer *et al*., 2008).

Montado grasslands show the highest content of OM (mean value higher than 4.5%), a clear indication that this content decreases with the intensification of land use, namely for lucerne cultivation, olive orchard and vineyard (mean value lower than 2.5%). The differences are statistically significant according to Duncan's test (Table 10). The low values of OM in those intensive land uses can be explained by soil mobilizations/tillage and irrigation systems. Other studies suggest that OM is higher in no-tillage soils compared to minimum tillage (Celik, 2005; Islam and Weil, 2000), since tillage mixes the subsoil with topsoil, inducing OM decomposition. Also, nutrients are easily leached by irrigation and the surface becomes poor in nutrients (Al-Kaisi and Licht, 2005). As for OM, then, the highest values of N nutrient occur in the Montado areas and the lowest values in lucerne cultivation, olive orchard and vineyards, which is connected to the plough/tillage practices that are frequently employed in these last land uses, while in the Montado grassland cattle enriches the soil.

HCsat values were greater in the Montado at the "Herdade do Roncão" farm (15.11 cm/h). Along with its sandy texture, these extensive agro-silvo pastoral systems (Montado) had been maintained with natural vegetation for about 10 years and, according to Alvarez and Steinbach (2009), the aggregation stability and water infiltration are higher in soils without mobilization systems. On "Herdade dos Gregos" and "Herdade do Pico", the HCsat values are lower for all the land uses, which can be explained by soil type, which corresponds to Luvisols (see sub-section 2.1.2, Figure 2) with clay-enriched subsoil, although it also can be a direct consequence of soil degradation and compaction resulting from repeated tillage operations (on lucerne cultivation, olive orchard and vineyard), overgrazing or the result of fire related events (which happened on montado grasslands in "Herdade dos Gregos"). Pagliai *et al.* (2004) declared tillage as being responsible for creating conditions that lead to soil compaction and Savadogo *et al.* (2007), as well as fire and intensive grazing.

As a result from the variations on soil properties induced not only by natural soil variability but also by different land use and management (anthropogenic factors), soil erodibility (K factor) changes. Comparing the different land uses, it is perceptible that K factor increased with the intensification of cultivation, with lowest values for Montado grasslands (0.023 and 0.021 t ha h ha^{-1} MJ^{-1} mm^{-1} for “H. Roncão” and “H. Gregos”, respectively) and highest for lucerne cultivation (0.039 t ha h ha^{-1} MJ^{-1} mm^{-1}), olive tree orchard (0.038 t ha h ha^{-1} MJ^{-1} mm^{-1}) and vineyard (0.029 t ha h ha^{-1} MJ^{-1} mm^{-1}). Other investigations have found similar results, showing that the mobilizations and the removal of permanent vegetation, decrease of OM, reduction of soil aggregation and porosity, caused by intensive cultivation, contribute to decrease of soil erodibility (Evrendilek *et al.*, 2004).

3.1.2.2.<u>Spatial distribution/ Geostatistics</u>

Through interpolation maps obtained with geostatistics it is possible to understand the spatial distribution of different soil properties and soil erodibility. The geostatistic parameters used for cross-validation in Ordinary Kriging (OK) are presented in Table 10, namely the nugget, sill, mean error (ME) and the root-mean-square standardized error (RMSSE).

Most of the soil properties were best fitted with an Exponential model. The nugget to sill ratio is used to examine spatial dependence of soil properties according Cambardella *et al.* (1994) (as described on the sub-section 2.3.2.2). As shown in Table 10 the ratio values indicate, for almost properties in different land uses, the presence of high to moderate spatial dependence (nugget to sill ratio lower than 0.75). There are only some soil properties on “Herdade do Roncão” (silt and sand content) and “Herdade do Pico” (clay content) with weak spatial dependence (nugget to sill ratio higher than 0.75)

The variability of spatial soil properties can be influenced by natural factors (as particle-size composition and topography) and anthropogenic factors (as land cover or management practices) (Tesfahunegn *et al.*, 2011). Sometimes, the effect of some factors is at least one order of magnitude greater (as topography or soil type) than the land use.

As mentioned, trend analysis was performed to study the existence of directional trends caused by these factors with large scale of variation. More detail about trend removal was published in Ferreira *et al.* (2015) (Appendix IV).

The interpolation maps of K factor for all land uses are presented in Figure 9.

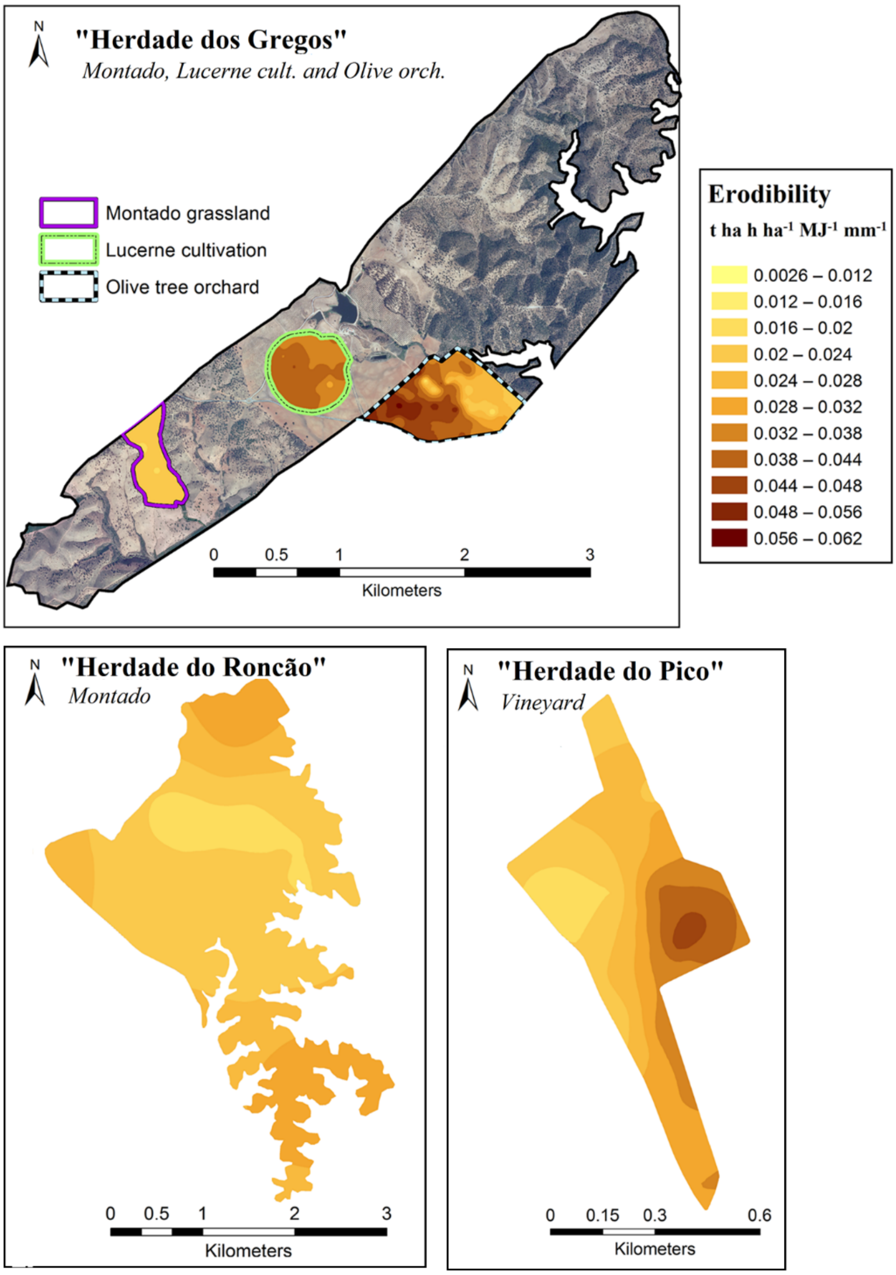

Figure 9 - Distribution maps of predicted soil erodibility factor (K) for each experimental area.

When looking at soil erodibility distribution, not only are the highest values on the cultivated field (lucerne cultivation, olive orchard and vineyard) present, but it is also worth mentioning that K is more homogeneous on the Montado areas when compared to cultivated areas. On the Montado areas, the spatial variability is mainly associated with natural (intrinsic) influences (such as texture) being soil properties and associated K factor being more homogeneous. In the cultivated fields, spatial variability is more dependent from not homogenous anthropogenic causes such as fertilization and irrigation rates and tillage/plough procedures.

The interpolation maps for some specific soil properties studied in “H. Roncão” e “H. Gregos” have been published in Ferreira and Panagopoulos (2014) (Appendix III) and Ferreira *et al.* (2015) (Appendix IV). By comparing K factor maps with soil property maps, it is easier to find the origin (the reason) of the highest susceptibility to erosion (high K factor values) in a specific point. Connections between K factor and soil properties (OM, N, VFS and MSilt [silt+VFS]) distributions were confirmed by observing the maps. The OM and N distributions presented by Ferreira *et al.* (2015) in the cultivation fields can be useful to identify inadequate management practices (*e.g.,* inadequate fertilization rates, tillage, irrigation rates, seed rates, etc.). On the other hand, it is essential to know soil property distribution when applying specific land uses and land management practices.

3.1.2.3.HJ-Biplot – “Herdade dos Gregos”

As previously explained, the HJ-Biplot is used to better understand the effects of land uses and specific practices on soil properties through the analysis of correlations between samples and between properties, as well as between each. This presents us with the results and discussion of the HJ-Biplot analysis done for “Herdade dos Gregos”.

The HJ-Biplot representation matrix of soil properties (only for land uses on “Herdade dos Gregos”) is available in Figure 10. It was observed that the dominant axis (axis 1) takes 35.83% of the total inertia (information) of the system. With both dimensions, an accumulative inertia of 61.04% was achieved.

Regarding this graphic representation, it was firstly noticeable that samples were grouped according to different land uses. The Montado samples were close to OM, N and Clay vectors, showing their preponderance to be a characterization of these variables (highest values of these properties correspond to Montado). The lucerne cultivation samples were important to describe the pH and Silt content. On the other hand, the Olive samples were more disperse but related to EC, Permeability class, Sand, VFS and K.

The variables demonstrating a great positive correlation between them were OM and N, as previously noticed. Clay and Silt were also positively correlated, but negatively correlated with sand as expected, because soils with more sand have less clay and/or silt.

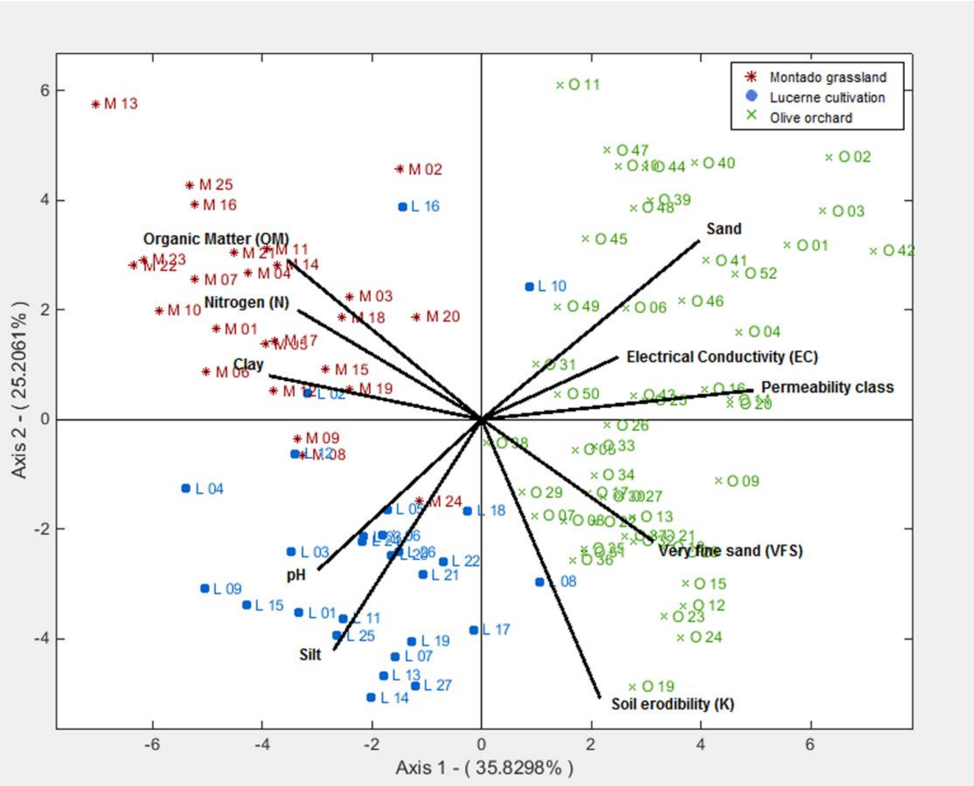

Figure 10 - The HJ-biplot representation matrix of soil samples and studied variables.

Through the matrix representation it was detected that soils with more sand have higher EC (olive orchard), although EC normally increases with the percentage of clay. This may be explained by the addition of fertilizers, as previously discussed, that can contribute to an EC increase. These results for EC show low variability between land uses, revealing a low cation exchange capacity (CEC) of these soils. This is frequently caused by intensive soil mobilization (Paz-González *et al.*, 2000).

Permeability class increases as the HCsat decreases, as defined by Renard *et al.* (1997). So, contrary to what was expected, the more sandy soils in this study (occurring in the Olive orchard) have less saturated hydraulic conductivity (high permeability class). This can be explained by a clay-enriched sub-layer under the sandy loam layer or/and by the soil compaction/degradation processes. The soil compaction and degradation can be related to repeated plow operations to reduce shrubs between olive rows and irrigation (Pagliai *et al.*, 2004). This permeability decrease in the Olive orchard was correlated with the increase in K factor.

Nevertheless, the properties more positively correlated with K factor were very fine sand (VFS) and silt; this is due to the susceptibility of these particles to erosion since they can be easily detached and transported by water (Morgan, 2005). The OM and N content were negatively correlated with K and permeability. The higher OM reduces the susceptibility of the soil to detachment and increases infiltration (Bronick and Lal, 2005). The nitrogen (N) content is not used to estimate K; however, especially for soils without fertilization, the existent N is mostly associated to OM. Nevertheless, nutrients decrease in soils that are more erodible, according to the studied literature (Tesfahunegn *et al.*, 2011). The clay content also shows a negative correlation with K factor, as expected (Renard *et al.*, 1997).

Figure 11 shows the representation of the hierarchical clusters. Using HJ-Biplot methodology and the aggregation tool ward, 3 clusters were obtained.

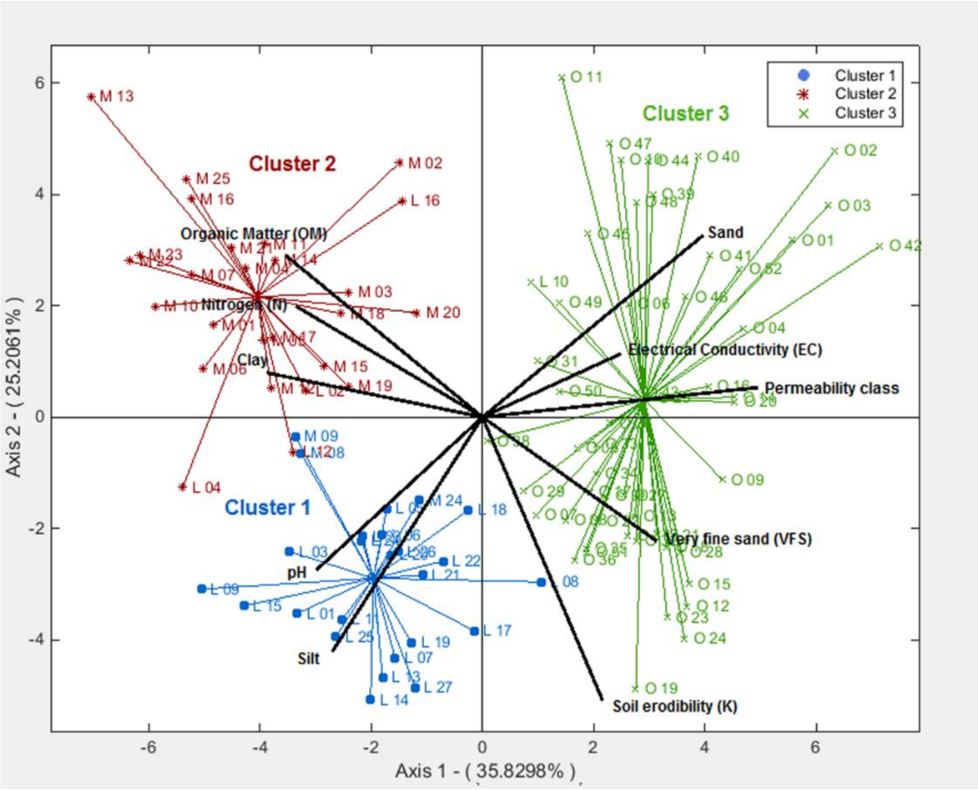

Figure 11 - Hierarchical clusters representation of soil samples and studied variables.

The samples were grouped by land uses (that were already detected by the matrix representation in Figure 10). Cluster 1 is represented by a majority of samples from Lucerne, Cluster 2 by samples from Montado and Cluster 3 by samples from the Olive orchard. This was explained by the effect of different management practices, vegetation cover and local soil characteristics, as discussed. Some samples in each land use had different values (higher or lower than the majority) and were grouped in a different cluster. By identifying the location of the sample, the cause of displacement can be studied and can help improve land management practices.

Cluster analysis, then, is convenient to identify the effect of different land use and management on soil properties, and consequently on soil erosion. On the other hand, the cluster analysis could support the delineation of zones according to soil properties and subsequent erosion susceptibility for specific management proposes.

3.1.3. Slope Length and Steepness (LS factor)

As previously mentioned, in order to create these maps, a DEM was used to provide information on elevation and to estimate the slope in each area, as well as the flow accumulation used to compute LS factor (Equation 5). The elevation maps were presented in the characterization of each experimental area (in the sub-section 2.1.3). The slope map for “Herdade do Roncão” computed to obtain LS factor can be consulted on Ferreira and Panagopoulos (2014) (Appendix III). The montado grasslands have the highest altitude variation and the highest slopes, with a maximum slope for “Herdade do Roncão” of 29.4%.

The LS factor maps for each experimental study area are presented in Figure 12 and the respective mean values per land use are shown in Table 11. The highest LS factor values (displayed in blue on the map) occur for Montado grasslands (with a mean value of 1.28 and 1.86 for “Herdade do Roncão” and “Herdade dos Gregos” respectively) and for vineyard (with a mean value of 1.21). These higher values are mainly associated to great slopes as confirmed in Ferreira and Panagopoulos (2014) (Appendix III). The lucerne cultivation and olive orchard present the lowest LS factor values (lower than 1).

Through the LS factor maps, it’s easy to identify the most sensitive areas that require special attention in the implementation of some land uses, especially cultivation practices that make soil more susceptible to erosion (high erodibility) and with little vegetation cover. Increases in this factor can produce higher overland flow depth and velocity, and thus higher erosion (Van Remortel, 2004).

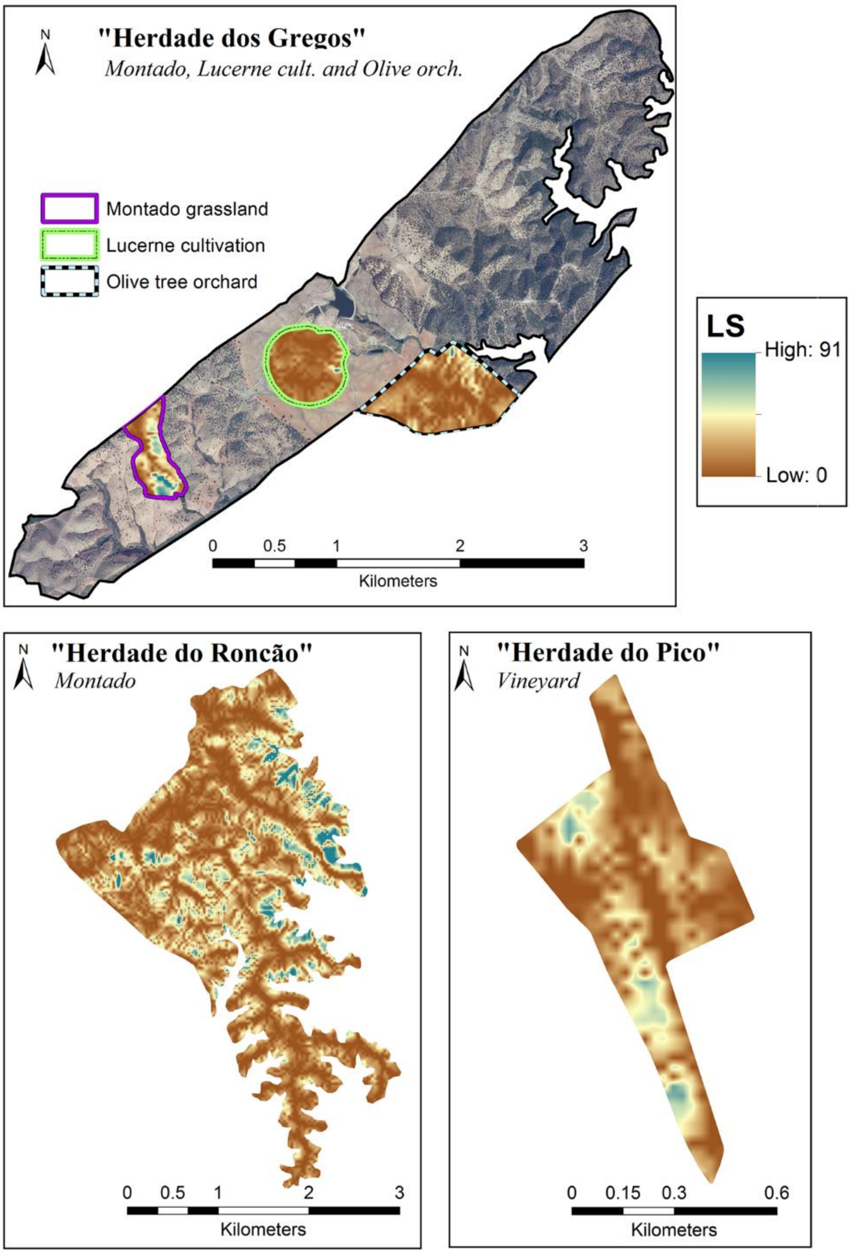

Figure 12 – Distribution maps of estimated LS factor for each experimental area.

Table 11 – Mean values of LS factor for each land use in the experimental areas (dimensionless).

	"H. Gregos"			"H. Roncão"	"H. Pico"
	Montado	**Lucerne cult.**	**Olive Orch.**	**Montado**	**Vineyard**
LS factor (mean)	1.86	0.49	0.81	1.28	1.21

3.1.4. Soil Vegetation Cover (C factor)

For each experimental area, and taking seasonality into account, the Normalized Difference Vegetation Index (NDVI) values were achieved through satellite imagery, and the associated C factor values were determined through Equation 7 (sub-section 2.2.4). NDVI and C factor maps were obtained for each season. The NDVI maps and the C factor maps are shown in Figure 13 and Figure 14, respectively. The respective mean values per season and for each land use are presented in Table 12.

The NDVI and corresponding C factor results demonstrated a clear influence of seasonality, a consequence of vegetation cover fluctuations throughout the year. Looking at the maps and their respective mean values, a negative correlation between NDVI and C factor values can be identified. The highest NDVI values, which suggest the greatest vegetation cover, result in lower C factor values, which, in turn, implies lower soil erosion.

Table 12 - Mean values of NDVI and C factor for each land use in the experimental areas (dimensionless values).

		"H. Gregos"			"H. Roncão"	"H. Pico"
		Montado	**Lucerne cult.**	**Olive Orch.**	**Montado**	**Vineyard**
Autumn	**NDVI**	0.127	0.673	0.156	0.110	0.152
	C factor	0.747	0.035	0.690	0.789	0.697
Winter	**NDVI**	0.484	0.600	0.348	0.455	0.193
	C factor	0.162	0.057	0.356	0.193	0.620
Spring	**NDVI**	0.241	0.226	0.169	0.222	0.224
	C factor	0.531	0.558	0.665	0.567	0.562
Summer	**NDVI**	0.107	0.138	0.110	0.102	0.138
	C factor	0.790	0.724	0.780	0.797	0.726

The lowest C factor values were found to occur mainly during the winter, when the NDVI values are higher, and the highest C factor values were estimated for summer and autumn, when the NDVI values are lower. This relation is particularly true for Montado grasslands, with natural vegetation cover (for grazing) that is more seasonal dependent, since its growth is conditioned by rainfall amounts and temperatures (Ferreira and Panagopoulos, 2012). Thus, on Montado grassland, water availability during the winter create conditions for vegetation growth, leading to high protection against rainfall-runoff erosivity (which leads to C factor levels being lower than 0.2). During the summer, high temperatures and dry conditions affect vegetation growth on this agro-silvo-pastoral system (which leads to C factor values being greater than 0.7). Despite the higher rainfall amounts in autumn, vegetation cover is still low after a long dry period, offering low protection against storms.

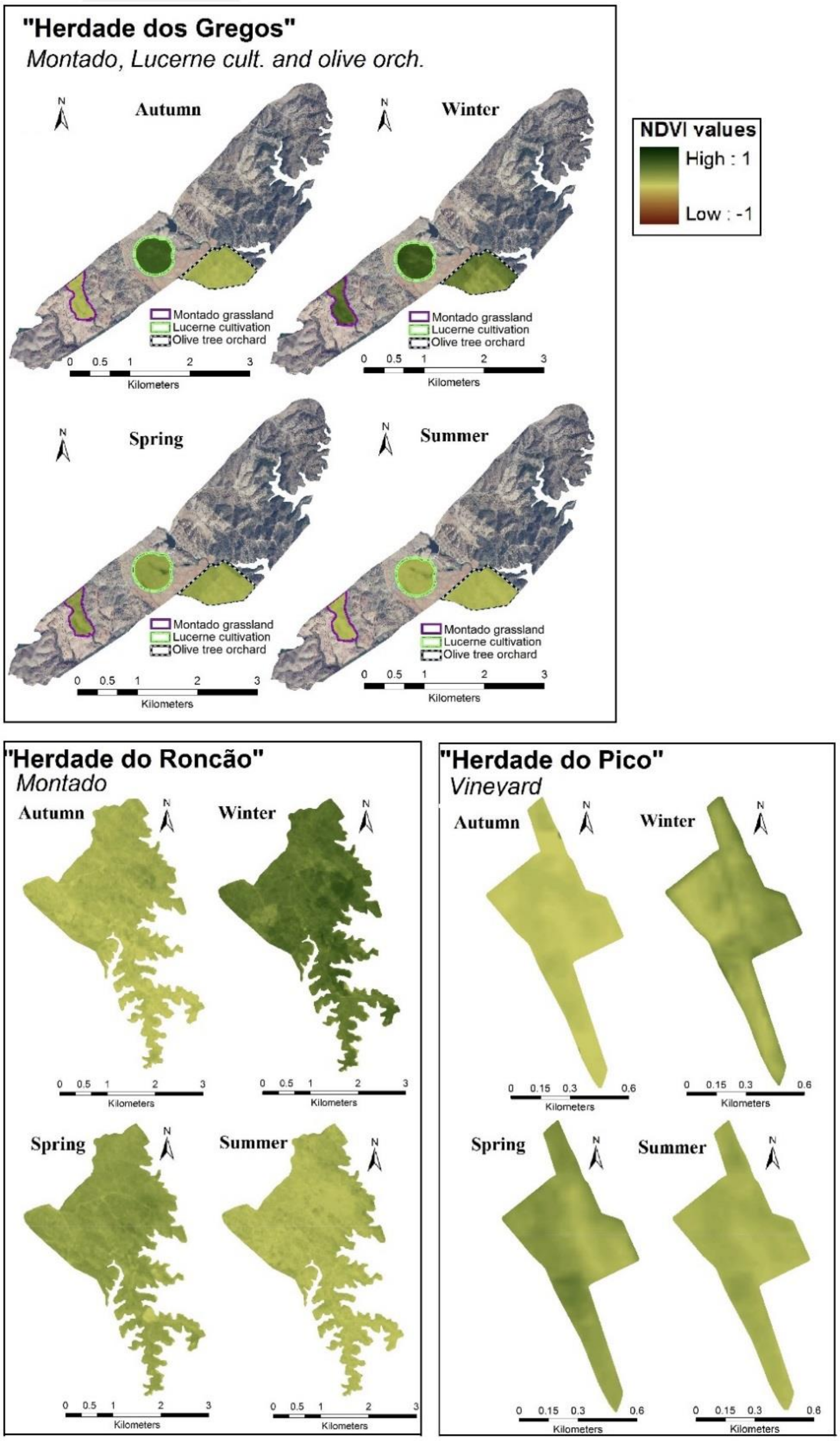

Figure 13 – Distribution maps of Normalized Difference Vegetation Index (NDVI) per season and for each experimental area.

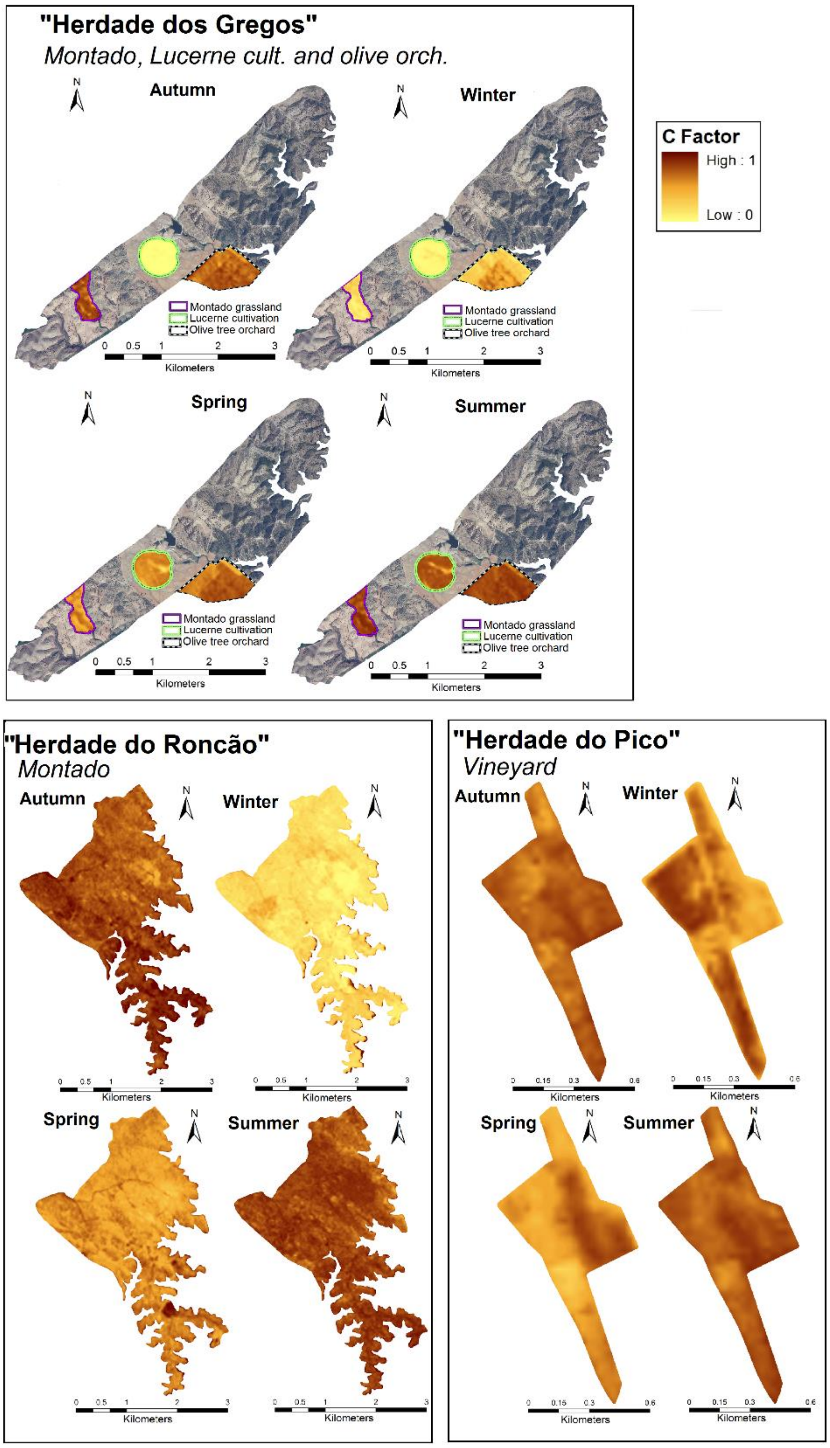

Figure 14 – Distribution maps of estimated C factor per season and for each experimental area.

On the other hand, in the more intensive land uses (lucerne cultivation, olive orchard and vineyard), vegetation cover is more reliant on farming practices. For lucerne cultivation, the vegetation cover (NDVI) is similarly high during winter; however, it is at its greatest in autumn. The high vegetation cover during the autumn season is an outcome of the use of irrigation systems, which leads to a C factor level that’s lower than 0.1. The olive tree orchard and particularly the vineyard have the low vegetation cover throughout the year (even during the winter months), which can explain constant soil mobilizations to decrease or eliminate natural vegetation between lines. On the vineyard these management practices lead to C factor values higher than 0.5 for all seasons.

3.1.5. Soil Erosion

RUSLE factors were integrated in the Raster Calculator under ArcGIS Spatial Analyst extension in order to calculate soil erosion rates and to get annual and seasonal maps for each experimental area. The annual and seasonal distribution maps obtained are shown in Figure 15. To facilitate comparing values between different land uses, the estimated means for soil erosion are found in Table 13.

For almost all land use (with the exception for lucerne cultivation), soil loss is estimated to be greater during the autumn season (more than 8 t/ha), adding up to more than 50% of annual soil erosion. A specific analysis of season contribution to the total annual soil erosion in the “Herdade do Roncão” is presented in next sub-section (3.1.6). As discussed earlier, during the autumn months soil erosivity reaches its peak due to heavy rainstorms; also, soil is particularly susceptible to this in the more natural land uses (since vegetation cover is still low to moderate after the summer months), because dry season carries with it high temperatures (Ferreira *et al.*, 2013).

For lucerne cultivation, despite higher soil erodibility (K), the predicted annual erosion is the lowest (3.5 t/ha) resulting from low slopes (LS) and from the protection of high vegetation cover (C). Through the irrigation system, the land owner in “Herdade dos Gregos” was able to have lucerne cultivation providing the maximum vegetation cover during the season with more rainfall-runoff erosivity (autumn). It can be considered a seasonal strategy for soil conservation.

On the other hand, the Montado in the “Herdade dos Gregos” showed the maximum value for annual soil erosion (19.1 t/ha), despite lower soil erodibility (K), a consequence of greater slopes (higher LS) and low vegetation cover since it had faced a fire related event 2-3 years before. The montado grassland in the “Herdade do Roncão” also presents high predicted soil erosion (15.0 t/ha), since the area has equally great slopes, and low vegetation growth because it was abandoned many years ago, being used only for grazing. Additionally in this area, some works have already started with the goal of implementing the tourism project mentioned before.

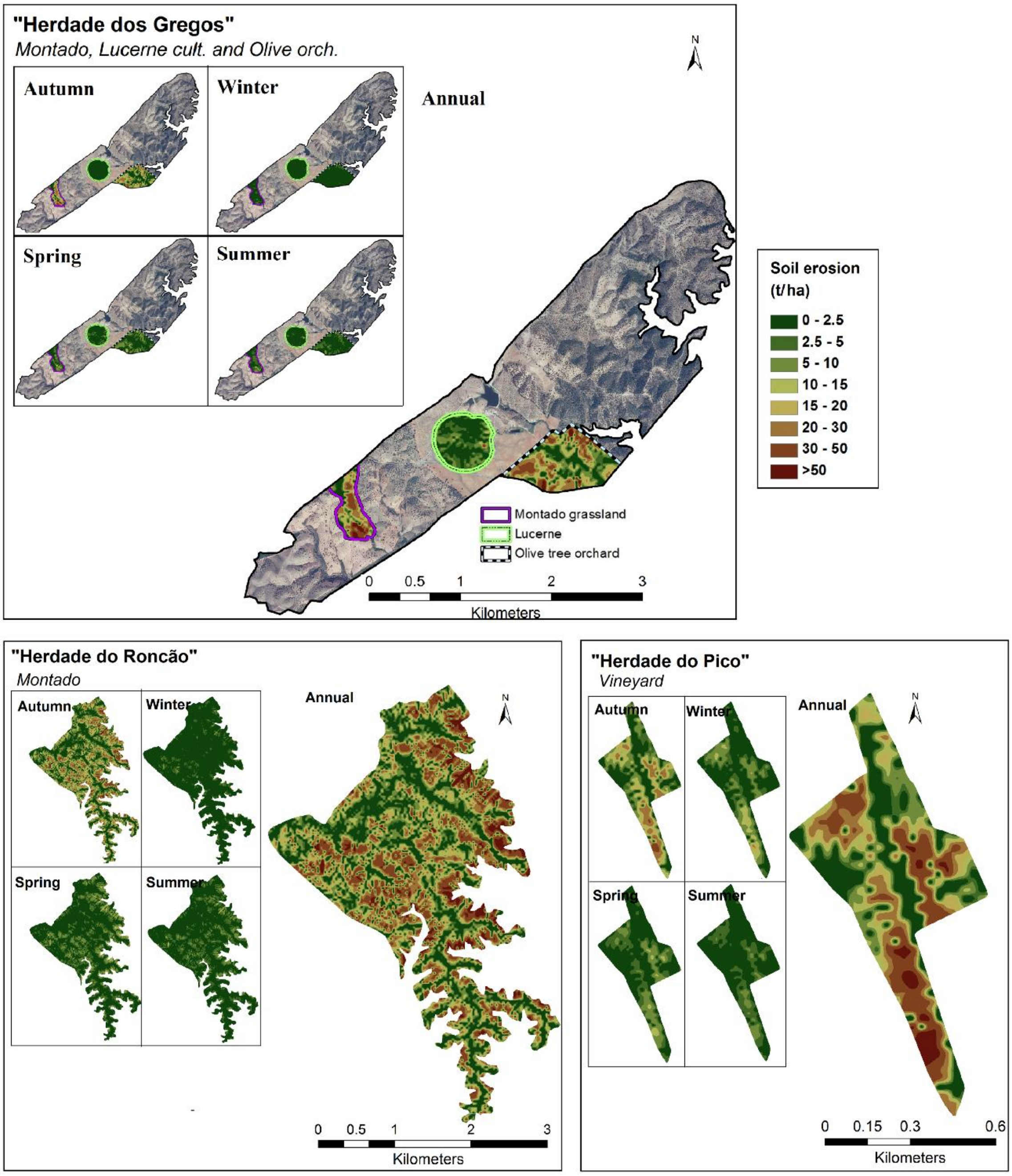

Figure 15 – Annual and seasonal distribution maps of soil erosion for each experimental area.

Despite its lower slopes (comparatively to montado grassland), the olive tree orchard and the vineyard land uses present high predicted values for soil erosion (15.2 t/ha and 16.9 t/ha, respectively). The main reason for this is essentially the high erodibility values in these land uses (section 3.1.2) and the low protection by vegetation even during the winter months due to vegetation removal between lines (section 3.1.4).

Table 13 – Annual and seasonal mean values of soil erosion for each land use in the experimental areas (t/ha).

Soil erosion (t/ha)	Montado	"H. Gregos" Lucerne cult.	Olive Orch.	"H. Roncão" Montado	"H. Pico" Vineyard
Autumn	12.770	0.447	9.438	9.730	8.823
Winter	1.175	0.303	0.826	1.031	3.440
Spring	2.736	1.615	3.068	2.438	2.778
Summer	2.267	1.117	1.908	1.820	1.851
Annual	**19.108**	**3.502**	**15.232**	**15.036**	**16.892**

The map distribution analysis allows the identification of sensitive areas ("hotspots") and confirms the seasonal variations mentioned earlier. Moreover, through these prediction maps, the topographic influence (LS factor) on soil erosion rates is evident.

The topography (LS factor) and the rainfall-runoff erosivity (R factor) are local characteristics, and are not influenced by land use. On these experimental areas only soil erodibility (K factor) and vegetation cover (C factor) factors are considered to be influenced by modifications on land use. A final analysis was then completed to compare the effect of land use, ignoring R and LS factors. Considering simply the ratio for KC factors, the following soil erosion susceptibility was obtained according land use: olive orchard>vineyard>montado>lucerne. Similar results were obtained by different authors, which observed higher soil erosion rates in orchards and vineyards compared to woodlands, scrubland or fire affected lands (Cerdà *et al.*, 2009; Kosmas *et al.*, 1997).

3.1.6. Soil Erosion Seasonality

As previously confirmed, soil erosion in the Mediterranean region is influenced by seasonality, associated with rainfall and vegetation fluctuations. This sub-section presents a seasonality analysis for "Herdade do Roncão", a montado grassland, where agricultural activities were abandoned and the intervention of human practices is insignificant. In this specific land use, fluctuations of vegetation (natural) are more dependent on season circumstances. The seasonality analysis allows for a better understanding of soil erosion and its seasonal contributions, the identification of critical periods, and the importance of seasonal variations in some RUSLE factors.

Figure 16 shows further evidence of the contribution of each season to the total annual soil erosion in the "Herdade do Roncão". Autumn was the season that most contributed to annual erosion (65%), followed by spring (16%) and summer (12%), with less impact during the winter months (7%). The maximum difference in soil erosion between seasons was 58%. This can be explained by the lag difference between rainfall-runoff erosivity and vegetation growth (shown more clearly in Figure 17). In autumn, the land was exposed to intensive rainfall occurrences after an extended dry and hot period that leaves soil less protected by vegetation.

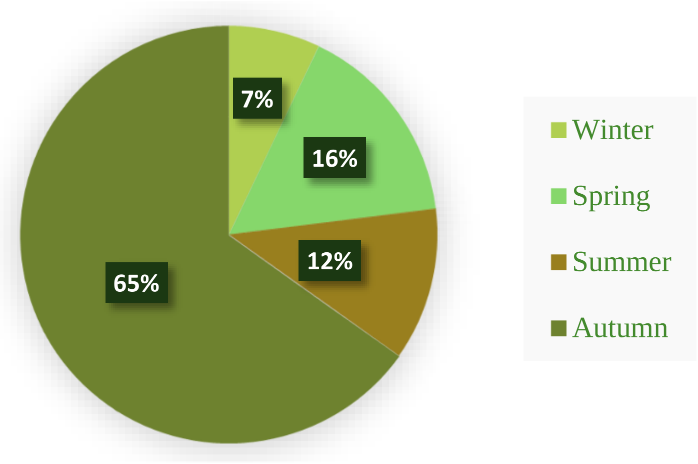

Figure 16 – Contribution of each season to annual soil erosion.

Figure 17 shows a temporal profile of soil erosion, associated NDVI values and rainfall-runoff erosivity. In the chart below, the mean values of NDVI were presented for each season (the mean value for the 3 year period), due to difficulty in obtaining for a monthly basis, as already explained. Rainfall-runoff erosivity and predicted erosion were presented per month.

As observed, there was a particularly great correlation between rainfall-runoff erosivity and soil loss. Vegetation (in this case analyzed through NDVI values) is also an influence, and it is noticeable in the first months of the year because the maximum vegetation cover contributes to decrease soil erosion, despite the fact that rainfall-runoff erosivity is still high. Throughout the spring and summer seasons, vegetation growth is decreasing, and so is rainfall-runoff erosivity. In October and November, the increase of rainfall-runoff erosivity associated with vegetation cover is still low, reflecting the highest erosion values, which exceeds 3.5 t ha^{-1} $month^{-1}$ in these months, for the experimental area.

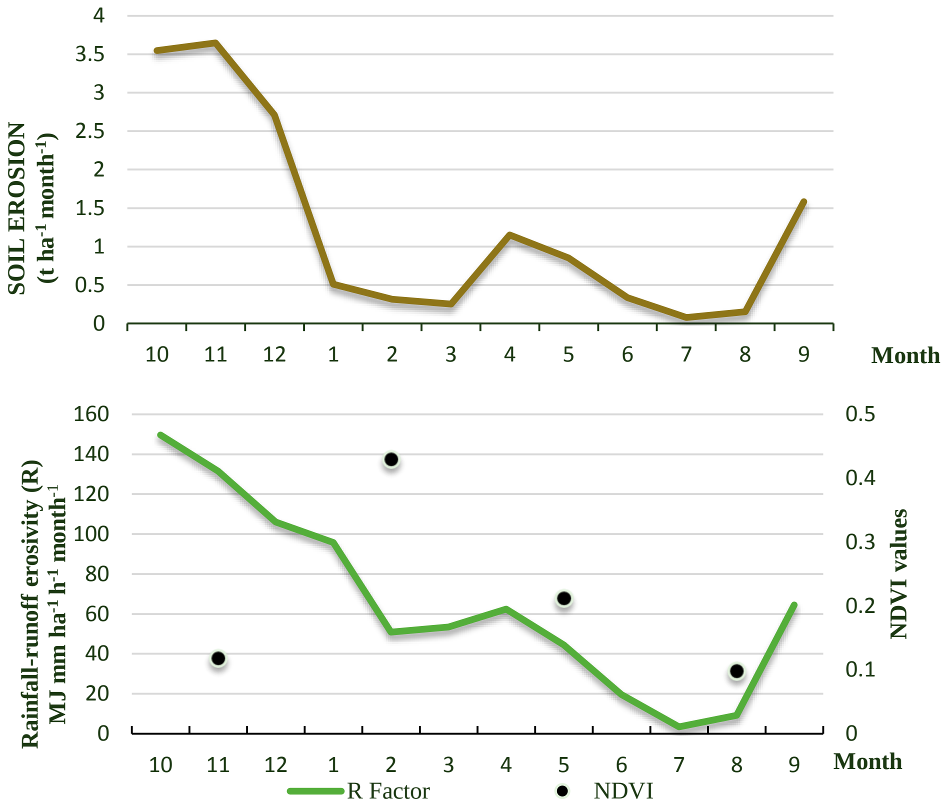

Figure 17 - Annual distribution of soil erosion versus NDVI values and rainfall-runoff erosivity.

Some studies have already assessed these seasonal patterns (Van Leeuwen and Sammons, 2003). However, these are especially notable in the Mediterranean regions that have particular climatic conditions (Van der Knijff *et al.*, 1999). Panagos *et al.* (2011), which obtained similar trends for the Strymonas river basin (SE Europe), registering the highest soil erosion during October and November, especially for forest, scrublands, and natural grasslands. Climate change could possibly intensify the differences between seasons and increase the frequency of heavy rainstorms throughout the year.

3.2. SOIL EROSION SCENARIOS

Previously to the forecasting and backcasting scenarios of soil erosion, an analysis of forecasted land-use/cover changes (LUCC) are presented to better understand in what way it will occur and their implications on soil erosion.

3.2.1. Forecasting Scenarios of Land-use/cover Change (LUCC)

The quantitative LUCC between 2000 and 2006 are presented in Table 14. As discussed, the Corine Land Cover (CLC) information for these years was used to estimate the probabilities of future LUCC change, using a Markov transition matrix. Between 2000 and 2006, in terms of area of land use types, the Alqueva landscape lost essentially broad-leaved forest (CLC class 311) equivalent to an area over 4000 ha in size and non-irrigated arable land (CLC class 211) of almost 3000 ha. The decrease of these land uses is essentially associated with the increase of transitional woodland/shrub (CLC class 324) land. Nevertheless, part of these losses were related with the gains in terms of land in agroforestry area (CLC class 244), permanently irrigated land (CLC class 212) and vineyards (CLC class 221).

Figure 18 illustrates land use according to Corine Land Cover in 2006 (assumed as the present situation) in order to compare with forecasted future scenarios.

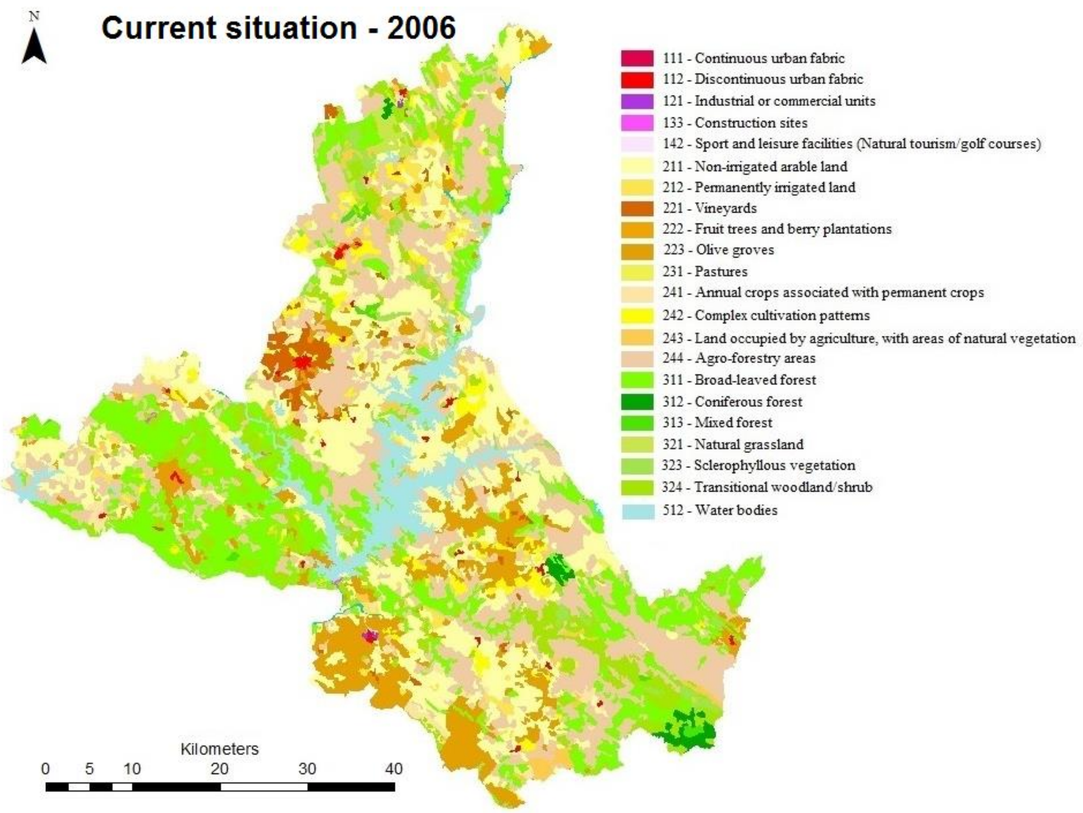

Figure 18 – Land use for the current situation (2006) according to Corine Land Cover.

In this scenario corresponding to current situation, most of the area was still covered by agroforestry area (CLC class 244), non-irrigated arable land (CLC class 211) and olive groves (CLC class 223), the typical land uses in the Alentejo region (Borges *et al.*, 2010; Jones *et al.*, 2011; Ferreira and Panagopoulos, 2015) until the challenges created by water availability from the Alqueva reservoir.

The forecasting LUCC scenarios for 2050 and 2100 were produced based on LUCC's transition matrix between the years of 2000 and 2006, different factors (considered decision-making agents) identified and its weights according to the different potential trajectories of land use change (production of biomass for bioenergy, agricultural intensification by means of irrigation, the increase of rural tourism and development of golf resorts, and climate change). All factors considered and their respective weights to obtain LUCC scenarios can be consulted in Samora-Arvela (2013). Figures 19 and 20 represent these forecasted LUCC scenarios for 2050 and 2100, respectively. The achieved changes are described quantitatively (in hectares) in Table 14.

Table 14 - LUCC in terms of area between 2000 and 2006, and for forecasting scenarios (2050 and 2100).

CLC Class	Past (ha)		Forecasting Scenarios (ha)			
	2000	2006	2050	Change 2006/ 2050	2100	Change 2006/ 2100
111	65.4	65.4	65.4	0	65.4	0
112	1300.4	1400	2396.3	996.3	2396.3	996.3
121	107.8	126.9	127.4	0.5	127.4	0.5
133	40.3	116.9	0	-116.9	0	-116.9
142	0	0	5824.4	5824.4	6777.6	6777.6
211	66513.7	64330.6	42226.2	-22104.4	42111.7	-22218.9
212	3248.2	3686.1	9976.1	6290	9821.7	6135.6
221	4587.1	4946	10587.3	5641.3	10449.2	5503.2
222	437.1	425.5	448.6	23.1	448.9	23.4
223	29934.5	30063.6	30218.2	154.6	30132.1	68.5
231	504.4	467.9	253.2	-214.7	253.2	-214.7
241	4570.5	4552.3	3132.1	-1420.2	3104.7	-1447.6
242	8868.4	8858.4	6229.7	-2628.7	6217.8	-2640.6
243	5977.8	5991.8	4623.2	-1368.6	4623.2	-1368.6
244	69306	70609.5	65144.6	-5464.9	65036.4	-5573.1
311	59374.6	55248.5	62207	6958.5	61935.9	6687.4
312	2648.1	2197.4	2753.1	555.7	2753.1	555.7
313	980.9	989.3	1654.8	665.5	1654.7	665.4
321	2036.1	2062.4	1423.6	-638.8	0	-2062.4
321[a]	0	0	131.7	131.7	1538.9	1538.9
323	5782.7	5239.5	2213.7	-3025.8	0	-5239.5
323[a]	0	0	26.1	26.1	2239.8	2239.8
324	16414.8	21320.8	25662.8	4342	0	-21320.8
324[a]	0	0	5186.7	5186.7	30824.2	30824.2
512[a]	0	0	186.6	186.6	186.6	186.6

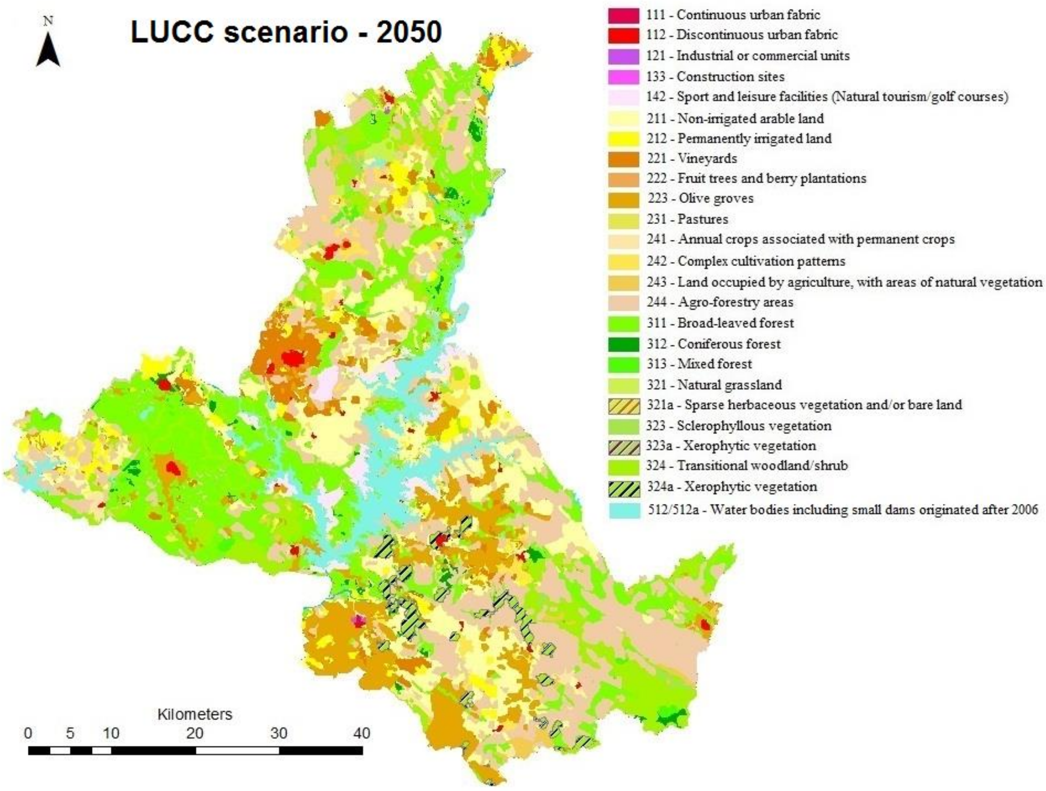

Figure 19 – Forecasting scenario of LUCC for 2050.

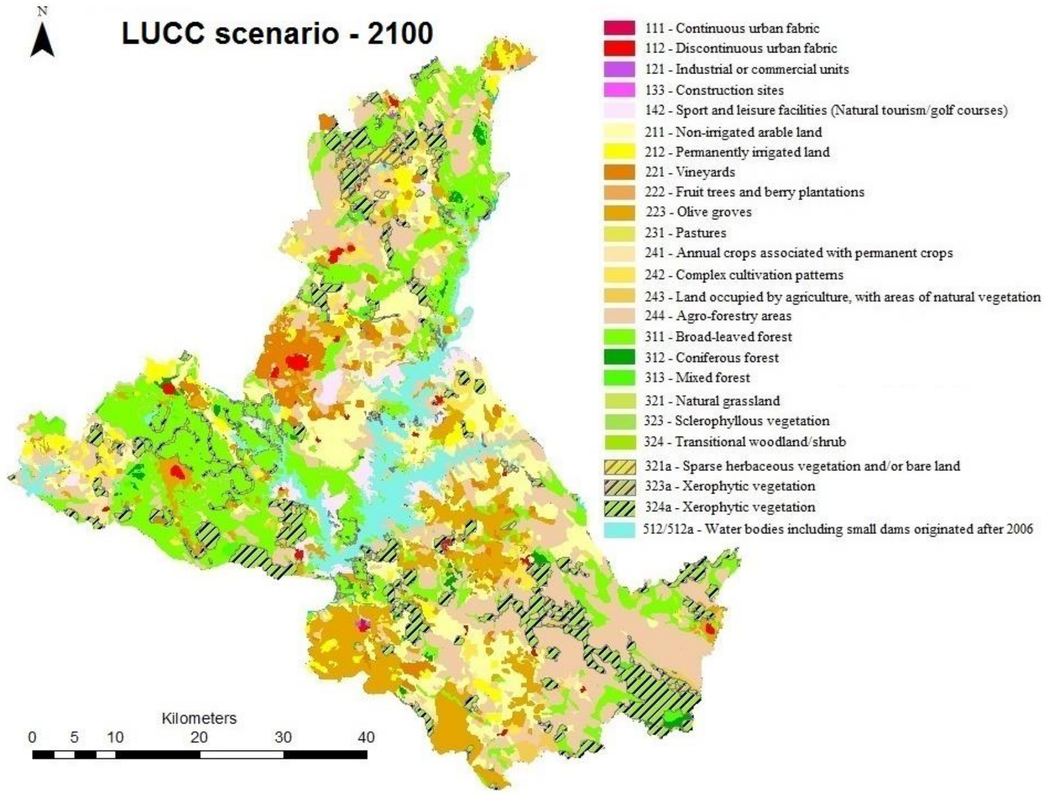

Figure 20 - Forecasting scenario of LUCC for 2100.

The main expected LUCC for 2050 and 2100 (when compared to 2006) are:

- an increase of discontinuous urban area (CLC class 112);
- a decrease of non-irrigated arable land (CLC class 211) and increase of permanently irrigated land (CLC class 212);
- an increase of sport and leisure facilities (natural tourism/golf courses) (CLC class 142);
- an increase of vineyards, olive groves, fruit trees and berry plantations (CLC classes 221,222 and 223);
- a decrease in the number of pastures, annual crops, complex cultivation patterns, agriculture with natural vegetation areas and agroforestry systems (CLC classes 231, 241, 242, 243 and 244);
- an increase in forested area (CLC classes 311, 312, 313);
- a decrease in the number of herbaceous natural vegetation (CLC class 321) and sclerophyllous vegetation (CLC class 323) and an increase of xerophytic vegetation and sparse vegetation (CLC class 321a, 323a and 324a);
- an increase in transitional shrub/ woodland (CLC class 324) by 2050, decreasing until 2100.

These results revealed the influence of water availability in the future of land uses on the Alqueva region, which will allow for the intensification of irrigated cultures. An increase of vineyards and olive groves using irrigation systems has already seen an increase in the region. For farmers, irrigation is a good strategy to diversify crop production and potentially increase income facing semi-arid conditions in order to overcome land desertification (Mira da Silva *et al.*, 2001). The expansion of natural tourism and golf areas is also a reality in the region, and some of the planned projects have already begun, thus taking advantage of the landscape created by the reservoir.

Transitional shrub/woodland is expected to increase between 2006 and 2050, especially due to abandonment of some agriculture with natural vegetation, pastures, agroforestry systems, and especially areas of non-irrigation as reported by Jones *et al.* (2011). The increase of forest area in 2050 and 2100 derives from the future expectation of forest plantation for bioenergy proposes due to irrigation systems and the proximity to a power station. Between 2050 and 2100, the transitional shrub/woodland, and, to a lesser extent, the forested area, are expected to decrease due to the conversion of some species to xerophytic vegetation, an expected result of climate change (Santos *et al.*, 2002) and continuous touristic development (golf course areas).

3.2.2. Forecasting Scenario of Soil Erosion

Based on the LUCC scenarios obtained, soil erosion was estimated using the RUSLE model in order to create scenarios of soil erosion for 2006 (Figure 21, present) and for 2100 (Figure 22, future). The soil erosion scenario for 2100, as mentioned, accounts for the influence of climate change on rainfall-runoff erosivity (R factor).

The estimated annual mean value of soil erosion for 2006 was 1.78 t ha^{-1}. Comparing the soil erosion scenario with the LUCC scenario for the same year, the highest soil loss values are mainly associated with olive grove plantations (CLC class 223, Figure 14) usually in terms of high altitudes (Sub-section 2.1.2 Figure 2), and consequently the LS factor. Traditionally, on olive tree orchards, tillage practice is used to control weeds and the soil was kept bare between trees, contributing to increase the susceptibility of soil erosion. Other small areas in 2006 presented the greatest soil erosion values as a result of construction activities where soil is frequently bare (a consequence of the new challenges created). As observed in the soil erosion maps in this study, the traditional montado grassland has some places with serious annual soil erosion (higher than 30 t ha^{-1}), mostly due to the combination of high slopes, reduced vegetation covers, and poor soils.

Comparing forecasting scenario for the year of 2100 with the present scenario in 2006, it is noticeable that the soil erosion rate is likely to increase in many places. The annual mean value for soil erosion for 2100 in the surrounding area of Alqueva reservoir is predicted to be 3.65 t ha^{-1}, representing an increment of more than 100% when compared with the mean value for 2006.

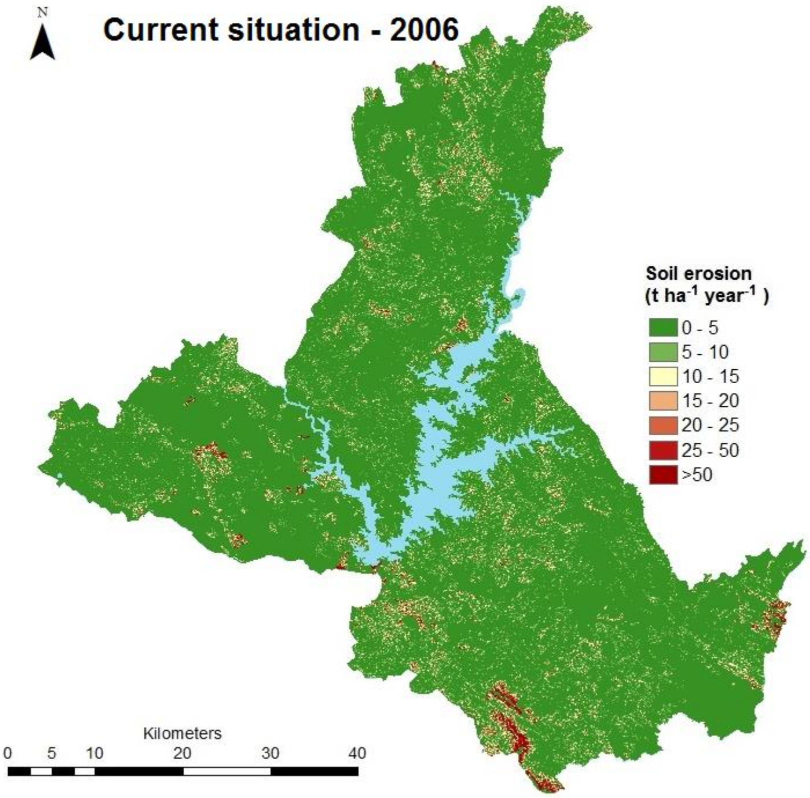

Figure 21 – Current situation for soil erosion (2006).

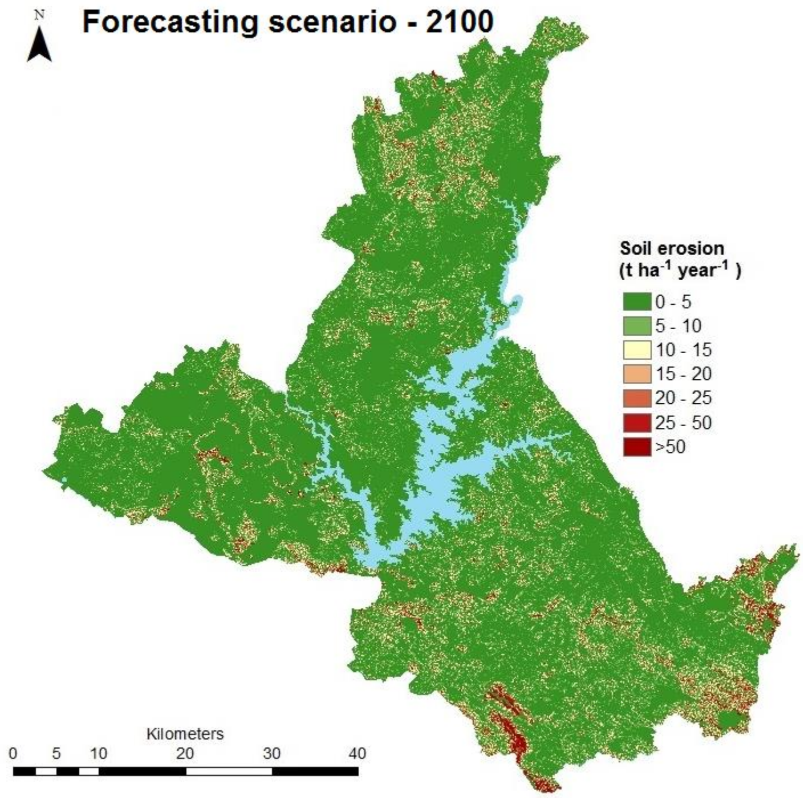

Figure 22 – Forecasting scenario of soil erosion for 2100 accounting R and C factor changes.

Soil erosion for 2100 is predicted to be higher than 50 t ha^{-1} year^{-1} for 5% of the study area, and these values are mainly found in the southeastern part. The intensification of the problem in that area is mainly associated with the change of transitional woodland/shrub and Sclerophyllous vegetation to Xerophytic vegetation cover and a decrease in forest density. Klooster (2003) mentions Xerophytic vegetation areas prone to suffer from sheet and gully erosion. On the other hand forest density is more likely to increase in the western part of the reservoir, which is connected to the potential of this area for biomass production (Samora-Arvela, 2013).

Additionally, the increase of some land uses such as vineyards, olive groves and other irrigated cultures contributes for the intensification of soil erosion in the future, since it is expected to reduce the annual mean value of vegetation cover (higher C factor as shown in Table 5, section 2.4) when compared to other land uses that have taken place before.

3.2.3. Backcasting Scenario of Soil erosion - Sustainable Land Management

The backcasting scenario for 2100 accounting for the implementation of sustainable land management (SLM) practices is represented in Figure 23. It is evident that the implementation of these common practices can effectively decrease soil erosion if applied, particularly in land uses with less protection from vegetation. From the scenario comparisons for 2100 (with and without SLM practices), one can establish the significance of SLM in the areas where the soil erosion rate is expected to be higher than 50 t ha^{-1} per year. The estimated mean value of annual soil erosion for 2100 with SLM is 2.27 t ha^{-1}, decreasing around 38% when compared to the 2100 scenario without SLM.

Despite the decrease of soil erosion susceptibility for 2100 accounting for SLM practices, the mean value of soil erosion is still greater than in the 2006 scenario (1.78 t ha^{-1}). This fact shows that the increase of rainfall-runoff erosivity (R factor), due to climate change, is likely to increase soil erosion even for better protected soils. Maeda *et al.* (2010) also studied the influence of climate change on soil erosion and established the importance of this knowledge for the development of optimal conservation practices.

For the studied backcasting scenario, the SLM practices were assumed specifically for the new land uses with low vegetation protection; however, facing climate changes, it is important to implement some other effective SLM even in the land uses considered less susceptible to erosion (specially pastures, annual crops, complex cultures and other agriculture with natural vegetation). Other different SLM practices have been applied on Mediterranean areas and the advantageous results were reported by WOCAT (World Overview of Conservation Approaches and Technologies) databases (Schwilch, 2012b).

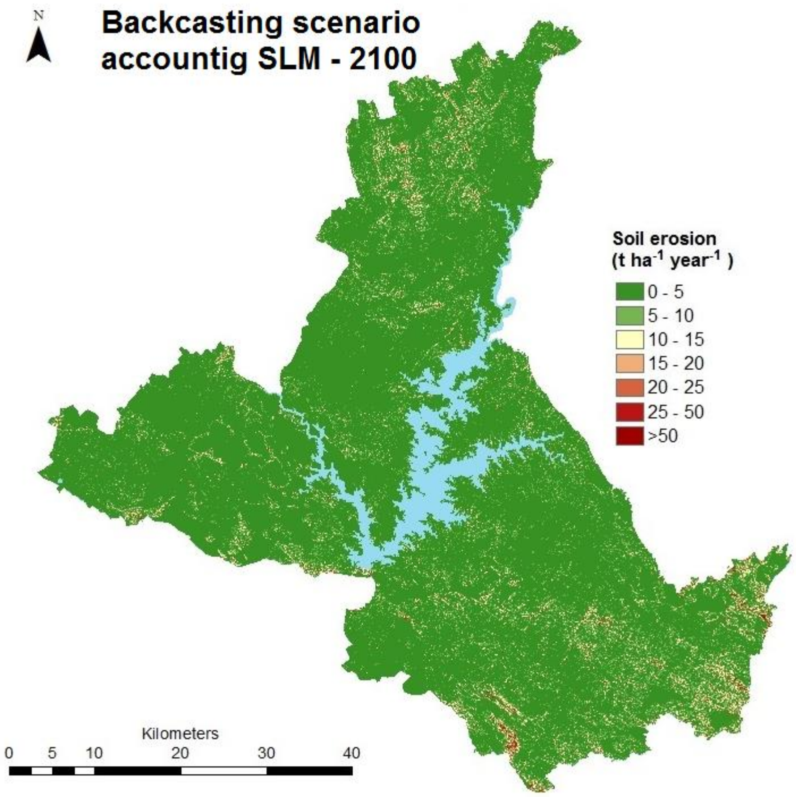

Figure 23 - Backcasting scenario of soil erosion for 2100 considering SLM practices (P factor change).

Studying future scenarios, it was not possible to include seasonal variations, since the C factor values for each CLC class correspond to an annual mean value. However, according to the seasonal analysis performed (sub-section 3.1.6), and as discussed by other researchers, soil erosion susceptibility is expected to be higher when accounting for seasonal variations on rainfall-runoff erosivity and vegetation cover than using mean annual values. In sum, in order to successfully select and define SLM practices for soil conservation, it is important to obtain and assure knowledge of local seasonal conditions. The SLM practices applied are efficient but not sufficient when facing the complexity of soil erosion in the region (see limitations of the study).

3.3. SEDIMENT DELIVERY

Through the equation presented on methodology (section 2.5) and knowing the size of study area (2 847 km^2) it was possible to estimate sediment delivery ratio (SDR) that is 0.175, as shown in Table 15. From the annual mean of soil erosion for each scenario previously presented, the sediment yield and the corresponding amount values were estimated. These sediment delivery results obtained are presented in Table 15.

For the current situation (2006), the predicted annual amount of sediments was 88 684 t $year^{-1}$, but an increase higher than 100% was predicted in the forecasting scenario for 2100 accounting R and C factor changes, particularly if no sustainable land management (SLM) practices took place in the study area (181 838 t $year^{-1}$). Despite the land use and climate changes considered, if common SLM practices are implemented (backcasting scenario), the annual sediment amount that goes into the reservoir can decrease about 38% (113 111 t $year^{-1}$) This reduction of sediments can be reproduced in a longer reservoir's life time and less water quality problems.

Table 15 – Sediment delivery estimations for different scenarios of soil erosion.

Scenario	Study Area (km^2)	Soil loss by RUSLE		SDR	Sediment Delivery	
		t ha^{-1}	t km^{-2}		Sediment yield (t km^{-2} $year^{-1}$)	Sediment amount (t $year^{-1}$)
Current situation (2006)	2 847	1.78	178	0.175	31.15	88 684
Forecasting scenario (2100)		3.65	365		63.87	181 838
Backcasting scenario (2100)		2.27	227		39.73	113 111

However, SDR, according to Jain and Kothyari (2000), is a spatial phenomenon that varies with watershed heterogeneity (such as topography, land use, soil characteristics, and rainfall-runoff erosivity), being important to discretize the watershed into sub-areas, each having similar homogeneous characteristics. The SDR-area relationship does not take it into account. Therefore, these estimations are an important indicator of sediment delivery amounts in the Alqueva watershed, and how different present and future scenarios can increase or decrease them.

3.4. SIMULATION MODEL AND DECISION SUPPORT SYSTEM

A dynamic simulation model was produced for the Alqueva region to easily estimate erosion in a huge set of situations, allowing for quick and clear interpretation of the data. It was associated with a decision support system (DSS) to suggest conservation practices according the land use. This is essential for local government land managers to be able to demonstrate the consequences of decisions to land owners.

The structure and running steps of the system created are represented summarily in Figure 24. The complete system procedure is executed in the following order (Cakula, 2012):

1. User opens the DSS and defines parameter values or uses previous scenarios in the graphical user interface (GUI),
2. Data is imported into Excel spreadsheet and values are transferred to the appropriate places;
3. Stella runs simulation model;
4. Simulation model imports data from the parameter Excel spreadsheet (connected with a persistent link) and runs the simulation;
5. Output data from the tables is automatically exported to an Excel spreadsheet (persistent link) with results;
6. Result spreadsheet is imported into GUI, the data is analyzed and important results are displayed;
7. Based on the results and on site conditions the DSS selects and displays corresponding recommendations;
8. Simulation results can be added to multiple scenario analysis in the DSS – they are stored in the temporary memory of the system;
9. Simulation can be run again by the user, incorporating recommendations to see the effects (repeat steps 2 through 6).

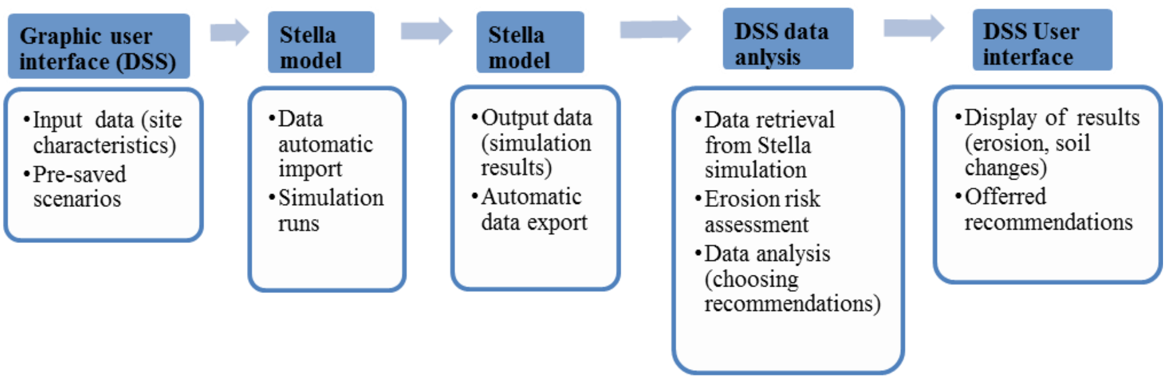

Figure 24 – Structure of the dynamic system (simulation model of soil erosion associated to a DSS) (Cakula, 2012).

A graphic user interface (GUI), is created as a DSS component, allowing for the definition of some parameters values according site characteristics. A screenshot of this GUI is shown in Figure 25. Information about area, soil properties, topography, land use practices and climate must be given by users to assess soil erosion risk on a specific site.

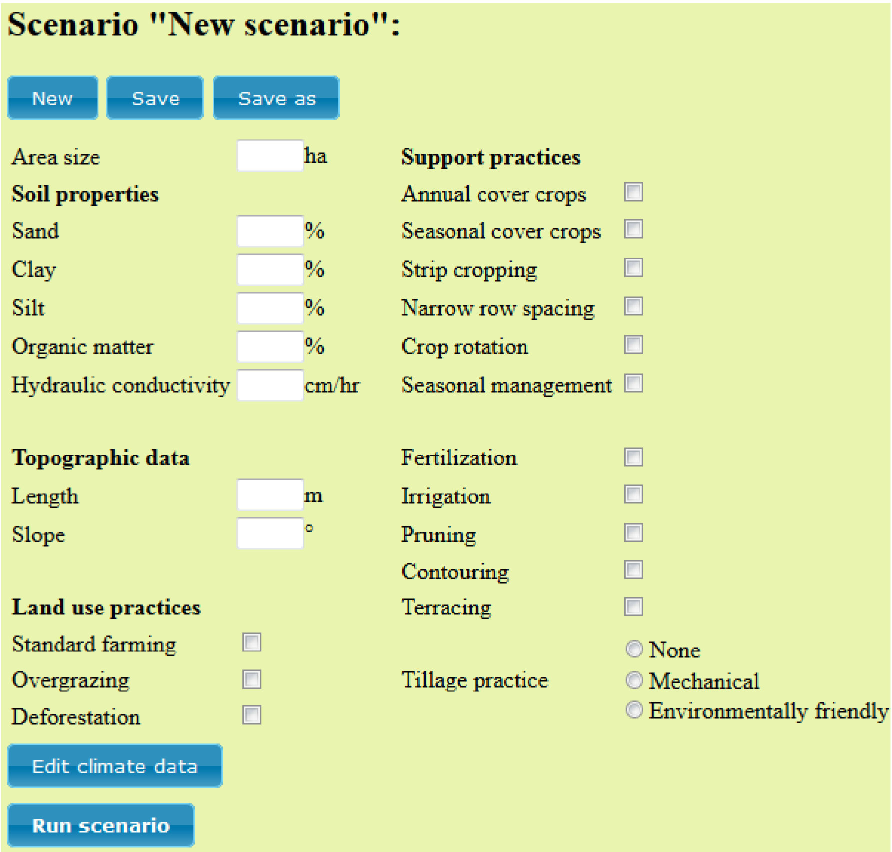

Figure 25 - Screenshots of graphic user interface to input site characteristics.

The simulation model created can be used in different scenarios of land use, including natural vegetation growth (the “base” scenario), standard farming and other possible agricultural scenarios. Moreover, it can be applied accounting for the presence of soil conservation practices (annual cover crops, seasonal cover crops, strip cropping, narrow row spacing, crop rotation, seasonal management, fertilization, irrigation, pruning, contouring, terracing), the application of three different tillage practices (none, mechanical and environmentally friendly) and two land mismanagement situations (overgrazing and deforestation).

As the scenario is run, the erosion at a specific site is modelled and assessed. The DSS shows the results, presenting the annual sediments per 1ha and the soil loss occurring throughout the year (graph), and it indicates a risk class. The risk of annual soil erosion was classified as: 0 to 15 t/ha is moderate; 15 to 45 t/ha is high; and above 45 t/ha is very high. An example of results analysis before and after recommendation implementation is presented on the Figure 26.

In the same user interface, the DSS offers recommendations which can reduce soil erosion (Figure 26). It is possible to rerun the simulation with implemented recommendations (one or more). A ‘base case’ scenario and the case with recommendations can be compared using both tables and a graph. Different scenarios with recommendations can also be compared.

Simulation modeling was successfully applied to predict erosion in a vast array of situations, thus saving time and resources. The system is not intended to make automated decisions in the place of decision-makers, but instead to improve their decision making process.

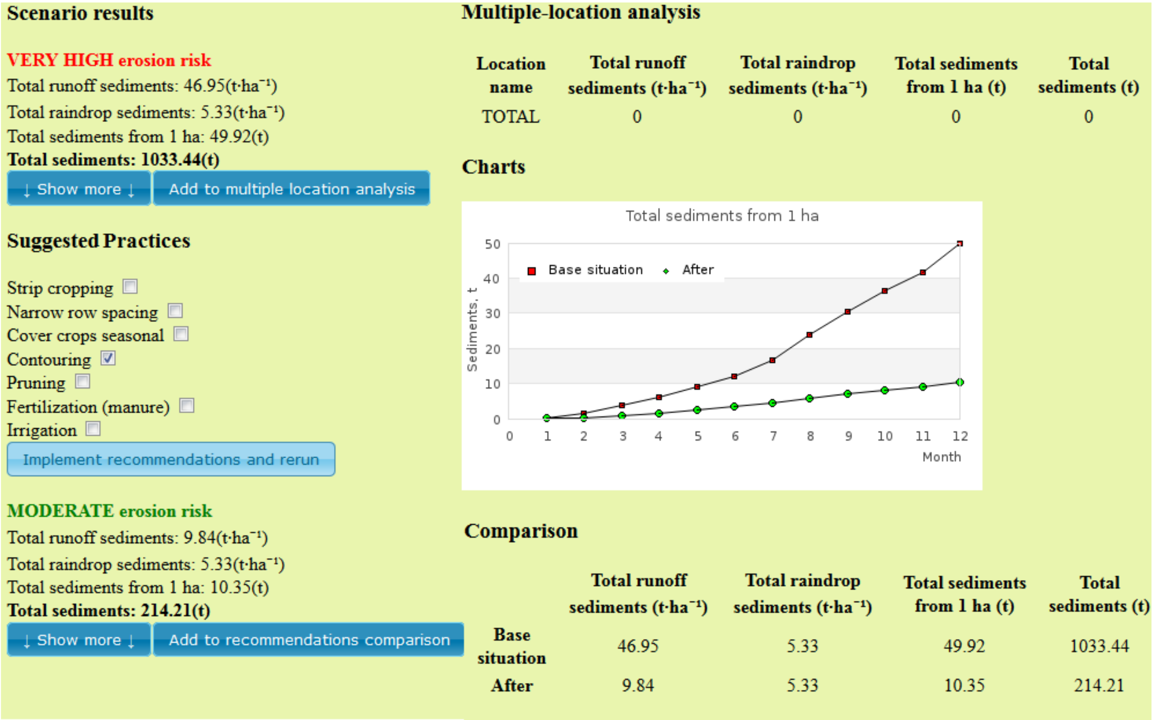

Figure 26 - Situation analysis before and after recommendation implementation (Cakula, 2012).

4. CONCLUSIONS

4.1. THE RESEARCH QUESTIONS

The investigation of soil erosion and its processes is of major importance regarding the management of natural resources. Soil erosion intensity determines the sustainability and productivity of agriculture, rangeland and forest systems, and the sedimentation and the water quality on reservoirs. An adequate sustainable planning of land use and management is essential to protect these natural resources and avoid unnecessary costs associated.

The environmental impact study of the Alqueva project gave insufficient significance to future soil erosion problems that may be created in the region due to climate change and land use change due to water availability. Consequently, sustainable land management strategies were not developed for the Alqueva region. According to the current investigation, soil erosion is a major threat in the Alqueva surrounding area. This semi-arid region is characterized by long dry periods followed by heavy bursts of intensive rainfall, falling on fragile, shallow and mistreated soils. Excessive soil erosion in the region resulting from non-sustainable land management could degrade the landscape and soil quality contributing to desertification in the long term, and also produce great amounts of sediments into the Alqueva reservoir decreasing its capacity.

The potential of geographic Information Systems (GIS) tools in combination with the Revised Universal Soil Loss Equation (RUSLE) prediction soil erosion model enabled the prediction of soil erosion rates and its spatial distribution under a variety of scenarios and conditions. RUSLE has shown to be practical and simple for field, farm and watershed scales. The input data used provide high resolution on each RUSLE factor studied and can be easily changed in the ArcGIS environment to study different areas with different land uses and management practices and local conditions. The analysis of each factor considered by the RUSLE equation and their spatial variability is fundamental to understand the main erosion inducers, as well as for the adequate planning of soil conservation measures, especially concerning agricultural practices.

The geostatistical techniques have shown to be useful for the estimation of some soil properties, K factor and R factor at unsampled locations, based on the sampled data, allowing for the analysis of its spatial variability. The HJ-Biplot shows an added value for spatial variability analysis and correlations of soil properties and K factor, and the simultaneous utilization with geostatistics allow for this information to be found spatially as well. Additionally, the clustering performed by HJ-biplot can be coupled in the map, which is useful as a land-management tool in the Alqueva area to identify areas with similar soil properties where similar land-management should be applied. The variability of soil properties is associated not only with local conditions (complex topographic landscape, soil type, etc.), but particularly with land use and cultural practices (tillage type, fertilizer rates, conservation measures, etc.). Therefore, in the surrounding area of the Alqueva reservoir, the ongoing change in land use and soil management practices can have a significant effect in chemical and physical soil properties. The analysis of specific different land uses, including the detailed study of soil properties, has shown that the intensification of cultivation (with tillage practices, fertilizations, constant sows and vegetation removal) associated with irrigation can effectively increase the susceptibility to soil erosion (soil erodibility). These practices are expected to decrease the amount of organic matter and nutrients and to reduce soil aggregation stability and porosity. As a result, this affects the soil erodibility index, intensifying the risk of erosion.

By updating the RUSLE model with the intra-annual variations of rainfall-runoff erosivity (R) and vegetation cover (C) factors, the soil erosion rates for each period were estimated, confirming the great seasonality that exists in this Mediterranean region. This study demonstrated the significance of incorporating temporal variability of some factors when modeling soil erosion. Soil erosion is a seasonal phenomenon greatly affected by changes in rainfall and vegetation cover during the year. When peak rainfall-runoff erosivity coincides with low vegetation cover, soil erosion risk is increased considerably. The greatest values of soil erosion are likely to occur during the autumn season, when rainfall erosivity reaches its maximum values and the vegetation (natural) is still low after the warmest and driest season. Keeping soil protected during this season can significantly decrease the susceptibility to soil erosion. It was confirmed in the lucerne cultivation that the effective application of irrigation systems can reduce soil erosion, since it allowed for greater vegetation cover during the season with the highest rainfall-runoff erosivity.

Therefore, understanding seasonal variations would be essential to delineate appropriate strategies of sustainable land management (SLM) for the watershed and for coping with the climate change challenges.

The prediction maps for each experimental area provide more detailed information for planning the future land use changes and for applying soil conservation practices. Additionally to the knowledge about critical periods/seasons for soil erosion, the distribution maps (including maps for each RUSLE factor and important elements) provide for a better understanding of local conditions, allowing for the identification of high-risk areas and their respective causes. It allows to find hot spots for risk of erosion and for the definition of priority areas regarding soil conservation. Also, the simple exploration of maps would allow to specialists to visualize the consequences of some local modifications in terms of land-use change and avoidance of sustainable land management practices.

In order to examine the significance of future soil erosion resulting from the expected land-use/cover changes (LUCC) and climate changes in the entire study area, it was necessary to understand the extent and conditions of this possible alterations. Multi-agent systems (MAS) was successfully applied in the entire Alqueva surrounding area to investigate forecasted LUCC scenarios taking into account different potential directions in the region. The LUCC associated with climate change is expected to greatly increase soil erosion. The creation of a backasting future scenario (accounting the application of some common soil conservation practices) has shown the importance of the implementation of SLM strategy in the region, facing the mentioned. However, despite the practices investigated having demonstrated the efficiency in reducing considerably the soil erosion, facing the extreme values expected, it is important to delineate site-specific measures accounting for seasonal patterns.

The dynamic simulation model created associated with a decision support system (DSS) has shown to be an advantageous tool to identify the susceptibility of an area to soil erosion in diverse conditions in the study area, and suggests specific measures to protect soil as well to decrease the risk of soil erosion. This tool can be operated locally by land managers in any point of the Alqueva region, changing the soil properties, topographic conditions, land use and management practices. According to the defined land use and predicted erosion risk, the system is able to suggest sustainable alternatives to reduce soil erosion. In the other hand, the current system can be upgraded by uploading new data, improving its helpfulness regarding land management.

Presently, many farmers lack knowledge on necessity and opportunities for sustainable land management. Information about local conditions, susceptibility to erosion and on appropriate land management should be made available to land owners in the area, not only to protect the resource soil but also the capacity of the reservoir to keep water for longer period. The exchange of information can be done using a DSS model; however the delineation of regional strategies involving different actors is essential.

Population growth, technology transfers, constant land use alterations and climate change will continue to exert pressure on the earth's resource along the world. As well as safeguarding the future of the reservoir and protect soils in the Alqueva region, this investigation aims to contribute more broadly to the soil erosion assessment and sustainable land management in the world. It provides important details and gives a better understanding of erosion processes, main driving forces and pressures, soil vulnerability and seasonal patterns under Mediterranean regions.

The information from this investigation is valuable for scientists around the world that aim at assessing the environmental impacts of erosion at the field or watershed scale. It reveals alternative approaches and methods, to assess and understand the current and future soil erosion rates, and their spatial and seasonal variability. Planning of soil conservation measures, especially concerning agricultural practices, requires a good knowledge of all these aspects. Additionally, this investigation shows a useful DSS tool, which can be adopted as a standard land management instrument in many areas around the world, exchanging the scientific knowledge with decision makers and farmers.

4.2. LIMITATIONS

The effectiveness of soil erosion modelling is a challenge considering the natural complexity of the processes and interactions, spatial heterogeneity and the lack of available data. All model applications are subjective, since the model is, by definition, a simplification of reality, and according to the data input choices, the model outcomes are influenced. These choices are not, however, unreasonable or irrational, and it doesn't result in predicting with poor accuracy. The quality of input data should be hand in hand with the complexity of the system (De Vente, 2009).

For this investigation, the input data for some factors was usually more detailed than the RUSLE requirements. The RUSLE is typically used to calculate total annual soil erosion, and the seasonal variations on rainfall-runoff erosivity (R factor) and vegetation cover (C factor) were overlooked, a limitation well known for this model; however, in the current work, the estimation of the aforementioned factors was updated to account for seasonal modifications in both.

Despite the temporal detail of input data, when studying the experimental areas, it was not possible to obtain more detail than season in regards to for vegetation cover. The satellite images available with good quality are only a few, since some of them display errors or a high percentage of cloud cover. One image was taken per season for a 3 year period, and the mean values of vegetation cover were used to estimate seasonal soil erosion. However, predictions obtained are equally useful for soil conservations strategies, since the same can be defined per season according to the season's average conditions.

The input data available is different according to the study scale. The scale influences the level of accuracy. In this investigation, an experimental scale at the level of farm ("Herdades") was used for a more precise characterization of soil erosion and its factors for different land uses. At this scale, it was possible to collect samples and characterize the soil erodibility in a more exact approach, and also to get satellite imagery to obtain vegetation cover and identify management practices. When moving to the scale of the entire study area (watershed scale, used for soil erosion scenarios based on LUCC), the level of accuracy and detail of input data decreases, since some generalizations were made to some of the factors:

- K factor: Soil Erodibility Map for Europe (resolution of 500 m) with lower resolution than the one obtained at the field scale;
- Land uses according CLC and vegetation cover based on tabled values; this doesn't allow for the study of seasonal soil erosion at the farm scale;
- P factor was given a constant value of "1" as for absence of conservation practices due to lack of data for the whole watershed.

However, the findings for a large scale are essential to define a sustainable watershed plan, and the outcomes from a field scale are important to validate and understand outputs from the model at a larger scale.

An important limitation of RUSLE model, as explained before, is the fact that this equation estimates only surface erosion amounts at a local scale and cannot be used to estimate the sediment yield for an entire study area. Local erosion from one site can be deposition to another site but without to reach the reservoir as sediment yield. Other models try to overcome this problem but overestimate erosion like the Unit Stream Power-based Erosion Deposition (USPED) model. To override this limitation, SDRs have been used to predict the amount of sediment transferred annually from the studied area into the reservoir. However, the transport of sediments and its deposition vary with watershed heterogeneity (such as topography, land use, soil characteristics, and rainfall-runoff erosivity), and the SDR-area relationship does not take it into account. Additionally, gully erosion, bank erosion and mass movements are not considered, even though these processes can have a large contribution to total sediment yield. The quantification of these processes should be included in future model development of sediment transport for more accurate and realistic values of sediment yield from a watershed. In sum, despite the low accuracy values, the sediment yield calculation provides for a better understanding of how future land uses and climate change can increase the sediments into the reservoir.

The developed dynamic simulation model with decision support system (DSS) is a very useful tool to assess soil erosion susceptibility in the region. However, not all of the land uses and conservation practices were considered due to the limited regional information at the initial phase of the study that the dynamic system was developed. It only considered natural vegetation and standard farming, failing to specify the differences on vegetation cover and phenology that were found at the posterior phase of the study as very significant for soil erosion assessment. Moreover, the spatial data was limited to the case study of “Herdade do Roncão”. As it is mentioned in the following section future research and recommendations, the dynamic simulation system can be further improved with the most recent data from this investigation and should be tested from the stakeholders.

4.3. FUTURE RESEARCH AND RECOMMENDATIONS

While there were many significant and interesting results taken from this study, there were many valuable aspects that presented themselves during this research and many different directions this research can take.

According to the results and limitations from this investigation, the following recommendations on future model development can be made:

- Adapting models (such as RUSLE based models) to account for seasonal changes particularly on the rainfall-runoff erosivity and vegetation cover factors, in order to increase their accuracy on modelling soil erosion. The RUSLE model is normally applied to assess annual soil loss and do not considers seasonal changes; however, strong seasonality was confirmed in this Mediterranean region and seasonal decision support system (DSS) will be helpful to take land management decisions and authorize land use change that take into consideration the seasonality of the above factors.
- Combining models to estimate the sediment yield from a watershed, accounting for different point sources of sediment (not only sheet and rill erosion but also gully erosion and mass movements), sediment transport and deposition.

In this study the most common land uses (montado grassland, vineyards, olive tree orchard and other crop cultivation) were investigated at an experimental scale; however, future research can include other possible land use like the emerging energy crops and new innovative management practices to improve the knowledge about soil erosion susceptibility in a vast array of situations.

As previously mentioned, facing some limitations of the dynamic simulation associated with DSS that was developed at the initial stage of this research, it requires to be updated with new data in order to become an adequate and valuable tool for soil conservation in the Alqueva reservoir and other similar Mediterranean areas. The created system should be improved expanding available spatial information for the entire study area, which can be imported from GIS sources, as it was developed in this investigation. It will allow the users to select a specific location from the map (with specific local conditions), introducing some information about land use and management and getting information on erosion susceptibility and alternative soil conservation scenarios (with graphs and figures as presented for the current system). Additionally, it should be updated with more common land uses in the Mediterranean region. The application of alternative soil conservation strategies should be investigated facing the new climate change challenges. Moreover, the dynamic system should be tested from the interested stakeholders (watershed managers, land owners, farmers, environmental non-governmental organizations [NGOs], World Overview of Conservation Approaches and Technologies [WOCAT]).

Therefore, and beside all the scientific knowledge, the collaboration with projects such as WOCAT can be useful for an adequate SLM regional strategy. The WOCAT is a global network that aims to promote improved decision-making on land management, sharing valuable knowledge with specialists and practitioners from all over the world (WOCAT 2007; Liniger and Schwilch 2002); allowing for knowledge exchange, WOCAT provides tools to identify situations in need of action and an SLM database with a full range of different case studies documented from all over the world (Schwilch *et al.*, 2012b). As future research, the WOCAT tools can be integrated with the knowledge from this investigation (map distributions) to analyze promising approaches and technologies, so as to provide SLM options for decision-makers. Furthermore, it can help through the estimation of the optimal alternatives for various land uses according to the specific characteristics of the terrain and the definition of soil conservation practices in possible, different land uses. Through the WOCAT tools, and through collaborations with both locals and governmental institutions, it will be possible to identify, assess, and select the best practices for any similar watershed, allowing for the delineation of seasonal strategies for different land use scenarios in order to reduce soil erosion rates. These strategies would be a part of a DSS for land degradation mitigation in the surrounding area of reservoirs.

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APPENDICES

Appendix I

List of all publications during PhD thesis work

Papers in international scientific journal with peer review

1. Panagopoulos, T., Ferreira, V. 2010. Erosion Risk map of Foupana river watershed in Algarve, Portugal. *WSEAS Transactions on Environment and Development*, 6 (9): 635-644.
2. Ferreira, V., Panagopoulos, T., Cakula, A. 2013. Prediction of seasonal soil erosion risk at the Alqueva Dam watershed, Portugal, Fresenius Environmental Bulletin, 22 (7a): 1997-2005. (Appendix II)
3. Ferreira, V., and Panagopoulos, T., 2014. Seasonality of soil erosion under Mediterranean conditions at the Alqueva dam watershed, Environmental Management, 54: 67-83. (Appendix III)
4. Ferreira, V., Panagopoulos, T., Andrade, R., Guerrero, C., and Loures, L., 2015. Spatial variability of soil properties and soil erodibility in the Alqueva dam watershed, Portugal. Solid Earth, 6: 383-392. (Appendix IV)
5. Panagopoulos, T., Cakula, A., Ferreira V., Arvela, A. 2015. Simulation model for predicting soil erosion in a large reservoir of southern Portugal. *International Journal of Sustainable Agricultural Management and Informatics* (in press).

   http://www.inderscience.com/info/ingeneral/forthcoming.php?jcode=ijsami
6. Ferreira, V., Samora Arvela, A., Panagopoulos, T. Soil erosion vulnerability under scenarios of land use change after the development of a large reservoir in a semi-arid area. *Journal of Environmental Planning and Management* (submitted).
7. Ferreira, V., and Panagopoulos, T., Cakula, A., Andrade, R., and Arvela, A., 2015. Predicting soil erosion after land use changes for irrigating agriculture in a large reservoir of southern Portugal. *Agriculture and Agricultural Science Procedia* (in press). (Appendix V)

Papers in national journal

1. Ferreira, V., Panagopoulos, T. 2012. Predicting Soil Erosion Risk at the Alqueva Dam Watershed, *Spatial and Organizational Dynamics – Discussion Papers: Landscape and Land-use Management*, 9: 60-80.

Papers published after peer review in conference proceedings:

1. Panagopoulos, T., Ferreira, V. 2010. Decision support system for erosion risk assessment. In: Mastorakis, N., Mladenov, V., Bojkovic, Z. (eds), *Latest trends on systems*, Volume I, Wseas Press, Athens, Greece, pp. 76-80. ISBN: 987-960-474-199-1.
2. Ferreira, V., Panagopoulos, T., 2010. Soil Erodibility Assessment at the Alqueva Dam Watershed, Portugal. In: Panagopoulos, T. (eds), *Advances in Climate Change, Global Warming, Biological Problems and Natural Hazards*. Faro, Portugal, pp. 112-118. ISBN: 978-960-474-247-9.
3. Ferreira, V., Panagopoulos, T. 2011. Predicting soil degradation under various land management scenarios at the Alqueva dam watershed. In: Kungolos, A.,

Karagiannidis, A., Aravossis, K., Samaras, P., Schramm, K.W. (eds), *3rd Int. Conf. on Environmental Management, engineering, Planning and Economics*, Skiathos, Greece, pp. 63. ISBN 978-960-6865-41-1.

4. Panagopuolos, T., Ferreira, V., Jesus, J., 2011. Determining the relation between soil erodibility and spatial variability of erosion properties using geostatistical techniques at the Alqueva reservoir area. In: Bojkovic, Z., Kacprzyk, J., Mastorakis, N., Mladenov, V., Revetria, R. (eds), *Proceeding of 10th WSEAS International Conference on Energy & environment*, WSEAS Press, Cambridge, UK, pp. 105-110. ISBN: 978-960-474-274-5.
5. Ferreira, V., Panagopoulos, T., 2011. Predicting Risk of Soil Erosion at the Alqueva Dam Watershed. In: Lekkas T.D. (eds), *Proceedings 12th International Conference on Environmental Science and Technology*, Rhodes, Greece, 8-10 September 2011, Vol B pp. 286-293. ISBN:978-960-7475-49-7.
6. Cakula, A., Ferreira, V., Panagopoulos, T., 2012. Dynamic Model of Soil Erosion and Sediment Deposit in Watersheds. In: Ramos, R.A.R., Straupe, I., Panagopoulos, T. (eds), *Recent Researches in Environment, Energy Systems and Sustainability*. Faro, Portugal, pp. 33-38, ISBN: 978-1-61804-088-6.
7. Arvela, S. A., Panagopoulos, T., Cakula, A., Ferreira, V., Azevedo, J., 2012. Analysis of landscape change following the construction of the Alqueva dam, southern Portugal – Approach and methods, In: Burley, J., Loures, L., Panagopoulos, T. (eds), Recent Researches in Environmental Science & Landscaping. Faro, Portugal, pp. 26-31. ISBN: 987-1-61804-090-9.
8. Panagopoulos, T., Andrade, R. Ferreira, V., Guerrero, C., 2012. Assessment of spatial variability of soil properties in areas under land use change due to the Alqueva dam construction. In Burley, J., Loures, L., Panagopoulos, T. (eds) *Recent Researches in Environmenal Science & Landscaping*, Faro, Portugal, pp. 26-31. ISBN: 987-1-61804-090-9.
9. Ferreira, V., Panagopoulos, T. 2012. Predicting soil erosion at the Alqueva dam watershed: seasonal variations. In: Ramos, R.A.R., Straupe, I., Panagopoulos, T. (eds), *Recent Researches in Environment, Energy Systems and Sustainability*, Faro, Portugal, pp. 99-104. ISBN: 987-1-61804-088-6.
10. Panagopoulos, T., Cakula, A., Ferreira, V. 2012. Soil erosion risk assessment at the watershed of Alqueva dam, Portugal. In: Katsifarakis, K.L., Theodossiou, N., Christodoulatos, C., Koutsospyros, A., Mallios, Z. (eds), *Protection and restoration of the environment XI*, Thessaloniki, Greece, pp. 1318-1327. ISBN: 978-960-99922-1-3.
11. Panagopoulos, T. Ferreira, V. 2013. Sustainable landscape management to decrease soil erosion risk and extend the life of dams. In: J.B. Burley (eds), *Modern Landscape Architecture*, WSEAS Press, Nanjing, China, pp: 44-48. ISBN: 978-960-474-355-1.
12. Ferreira, V., Panagopoulos, T. 2013. "Risco de erosão do solo na área envolvente à Barragem do Alqueva". In Borrego, C., Miranda, A., Aroja, L., Fidelis, T., Castro, E.A., Gomes, A.P. (eds), *10ª Conferência Nacional do Ambiente*, Aveiro, Portugal, pp. 602-607. ISBN: 978-989-98673-0-7.

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14. Panagopoulos, T., Ferreira, V., Karanikola, P. 2014. Evaluating landscape changes at the Alqueva reservoir. In: Liakopoulos, A., Kungolos, A., Christodoulatos, C., Koutsospyros, A. (eds), *Protection and Restoration of the Environment XII*, Skiathos, Greece, pp. 240. ISBN 978-960-88490-5-1.
15. Ferreira, V., Arvela, S.A., Panagopoulos, T. 2014. Scenarios of land use change and soil erosion risk at the Alqueva reservoir area. *The Proceedings of the 10th Congress of the Hellenic Geographical Society*, Thessaloniki, Greece.

Appendix II

Published paper:

Ferreira, V., Panagopoulos, T., and Cakula, A. 2013. Prediction of seasonal soil erosion risk at the Alqueva Dam watershed, Portugal, *Fresenius Environmental Bulletin*, 22 (7a): 1997-2005.

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PREDICTION OF SEASONAL SOIL EROSION RISK AT THE ALQUEVA DAM WATERSHED, PORTUGAL

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This study was presented at the Third International Conference on Environmental Management, Engineering, Planning and Economics (CEMEPE 2011) & SECOTOX Conference, 19- 24 June 2011 – Skiathos island, Greece.

ABSTRACT

Management of reservoirs is of major importance regarding the water supply in Portugal. The Alqueva reservoir constitutes the largest dam in Western Europe, however the capacity cannot be maintained due to a yearly deposition of sediment resulting in a silting up. The changes of climate and land use in the surrounding region could intensify this situation even more. This research integrates the Revised Universal Soil Loss Equation (RUSLE) with geostatistical techniques and a Geographic Information System (GIS) to model potential soil erosion in the Alqueva watershed. Modelling was used to develop the maps for soil erosion on smaller sub-watersheds of Alqueva, to help make conclusions for the total area. The results showed that soil erosion risk is higher during summer (with a mean of 38.1 t ha^{-1} $year^{-1}$) than during winter (with a mean of 14.1 t ha^{-1} $year^{-1}$). It can be explained by the fact that vegetation cover is low in summer and high in winter. Vegetation cover is clearly an important soil protecting factor. Furthermore the highest values for erosion are associated to the highest gradient slopes. Site-specific management methods that take seasonal variations into account are essential to decrease soil erosion. The erosion prediction maps could be used as a solid base to create a Decision Support System so as to provide specific procedures for decision-makers, according to local conditions. It will be important for promoting sustainability of the ecosystems, reducing the risk of erosion while consequently increasing lifetime of the Alqueva dam under various land management scenarios.

KEYWORDS: geostatistics; erosion; environmental management; planning.

1 INTRODUCTION

Soil erosion is an environmental and economic problem worldwide. The negative effects are related with the decline of soil productivity and the increase of runoff, which results in the acceleration of natural sedimentation in rivers and reservoirs reducing their storage capacity as well as life span [1, 2]. This problem has been accepted as sensitive both by climate and land use changes [3, 4]. Mediterranean environment may be particularly vulnerable to changes because of contrasted climate, low vegetation cover and specific poor soil characteristics [5].

During the last decades, several initiatives have assessed the risk of soil erosion. RUSLE is the most widely used empirical equation for estimating annual soil loss from agricultural watersheds [6], because it is simple, economic to use and can be implemented with limited data [7]. Geographic Information Systems (GIS) techniques and remote sensing have been successfully used in combination with these models because they make soil erosion estimation and its spatial distribution feasible with reasonable costs and accuracy [8-10]. Geostatistics have been used increasingly to generate prediction maps and to account for the local uncertainty [11-13].

Management of reservoirs is of major importance regarding the water supply in Portugal. The Alqueva dam surrounding region now has new challenges as traditional land-uses are changing rapidly due to water availability in this semi-arid region and therefore new risks are arising. Land-uses are expected to change through the intensification of agriculture and increase of irrigation area, bio-energy production and tourism development. Seasonal changes are verified in the literature which suggests that erosion risk may change according to intra-annual vegetation cover due to change in the traditional agro-silvo pastoral system management [11]. Therefore, the objective was to use the RUSLE model in combination with GIS and geostatistics and estimate the seasonal potential soil erosion risk at a case study area of the Alqueva reservoir region.

2 MATERIALS AND METHODS

2.1 Description of the study area

The Alqueva reservoir is located on the river Guadiana in Alentejo, a semiarid region in the south of Portugal

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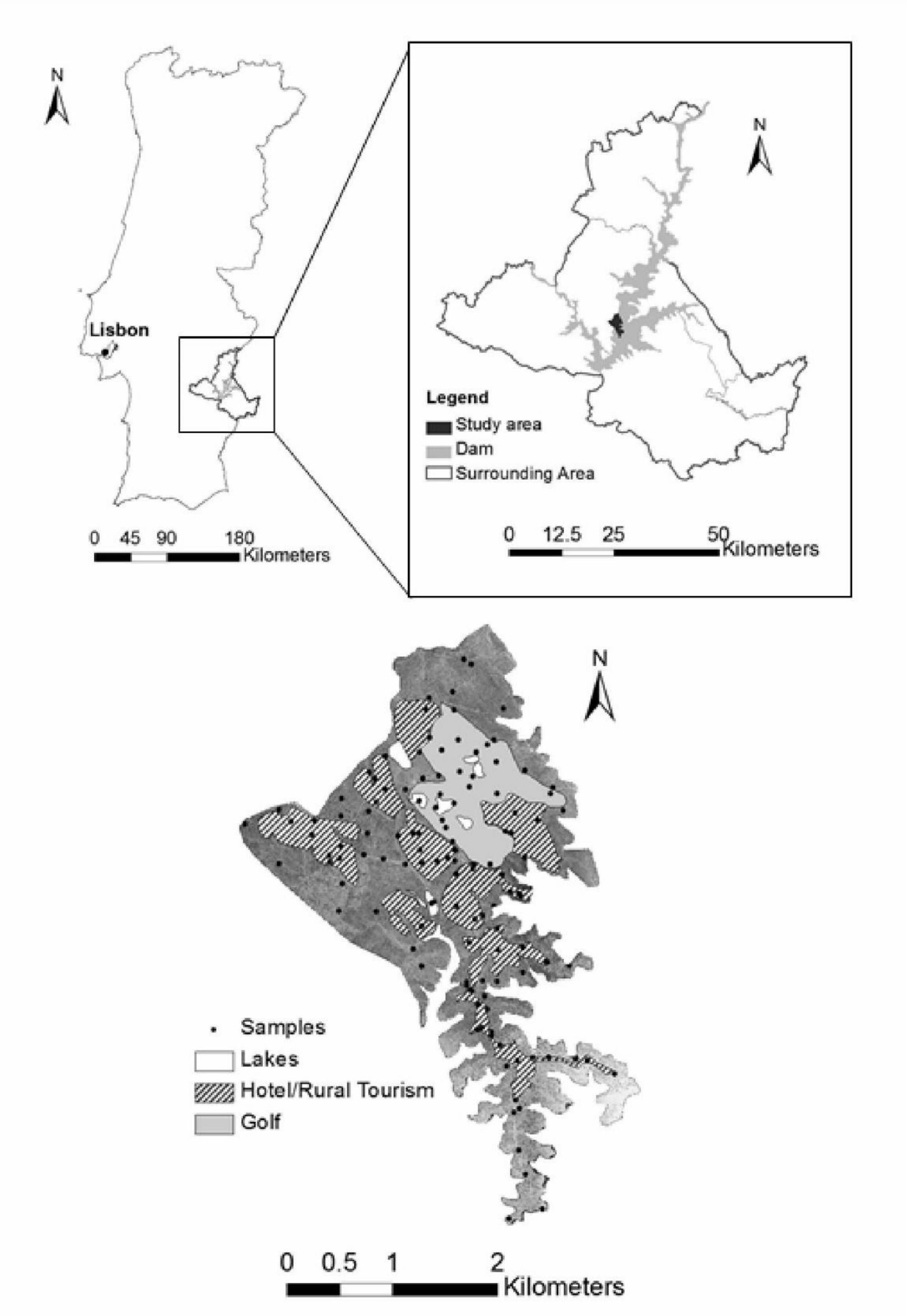

FIGURE 1 - Location of the study site.

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(8º30' W, 38º30' N). It covers an area of 250 km^2 (from which 35 km^2 are in Spain) and the total capacity of the reservoir is 4150 hm^3. The lake total shoreline is approximately 1100 km. It extends for 83 km and is considered the biggest in Western Europe [14], however its capacity cannot be maintained due to a yearly deposition of sediment resulting in silting up. An experimental research site (Figure 1) was chosen to study the potential risk of the soil erosion in this region.

The study area is in Herdade do Roncão beside the dam, near the Regengos de Monsaraz city. A tourism project will be implemented in this site. It has about 739 ha and the future land-uses are: a marina, a hotel and several golf areas. Currently, the typical local landscape is called "montado", an agro-silvo pastoral system. The main vegetative species in this system are *Quercus ilex L.*, *Quercus-suber L.*, *Oleaeuropaea L.*, cereals and fodder plants. The climate of the area is continental Mediterranean, with very hot and dry summers and mild winters [15]. For the last 7 years this area has not been cultivated, and the soil had been kept with natural vegetation as pasture.

2.2 RUSLE and data processing

In this study RUSLE was used to predict the average annual soil loss in the area before implementation of the tourism project changes actual conditions. RUSLE is defined as:

$$A=R\ K\ L\ S\ C\ P \quad \textbf{(1)}$$

where A= potential erosion (computed annual average soil loss in t ha^{-1} $year^{-1}$), R= rainfall and runoff factor, K= soil erodibility factor, LS= slope length and gradient factor, C= vegetation cover factor and P= vegetation control practice factor [6].

The rainfall-runoff erosivity factor (R) is generally known as one of the most important indicators for erosive potential of raindrops impact. It also reflects the potential of runoff generated by erosive storms. This R factor is the sum of erosive storm values EI30 occurring during a mean year, which result for the product of total storm energy (E) times the maximum 30 minute intensity (I30), where E is in MJ/ha and I30 is in mm/h [6]. Coutinho *et al.* [16] obtained an exponential relationship between rainfall and erosivity index. The equation was defined as:

$$EI30=0.33P-52.9 \quad \textbf{(2)}$$

where P is precipitation. In the study we used this relation with precipitation data from 137 meteorological stations, between 1991 and 2011, to estimate erosivity. An erosivity map for south of Portugal was created with geostatistical techniques.

Soil erodibility factor (K) represents the susceptibility of a soil to erode and the amount and rate of runoff, as measured under the standard unit plot condition. The unit plot condition is a continuously cultivated fallow plot, 72.6 ft (22.1 m) long with a slope of 9% [6]. In this study Kfactor was estimated experimentally using an algebraic approximation of the nomograph [6]:

$$K=[2.1\times10^{-4}(12\text{–}OM)M^{1.14}+3.25(s\text{–}2)+2.5(p\text{–}3)]/100 \quad \textbf{(3)}$$

where OM is organic matter, s is soil structure, and p is permeability class. M is the product of the primary particle size fractions (% Silt) × (%Silt + %Sand), where % Silt is percent modified silt (0.002-0.1 mm), and % Sand is percent modified sand (0.1-2 mm). K is expressed with U.S. units and division with the factor 7.59 will yield K values expressed in SI units of t ha h ha^{-1} $MJ^{-1}mm^{-1}$. A total of eighty-two (82) soil samples (0–20 cm depth) were collected and later analyzed in laboratory to obtain information about soil texture and content of organic matter. Permeability was analyzed in field using an infiltrometer. Permeability class and soil structure class was defined in accordance with Renard *et al.* [6]. Computed K factor values for each soil sample unit were added into GIS environment and a continuous surface representing the spatial distribution was created using geostatistics.

The factor LS indicates the impact of length and slope of terrain on soil erosion. Slope length (L) is the horizontal distance from the origin of overland flow to the point where the slope gradient decreases enough that deposition begins or runoff becomes concentrated in a demarcated channel. The slope steepness (S) shows the influence of slope gradient [6].To estimate this factor we created a Digital Elevation Model (DEM) in ArcGIS software [17] by digitizing contour lines from topographic maps. The combined LS factor was computed for the watershed by means of ArcGIS spatial analyst extension using DEM, following the equation (4), as proposed by Moore and Burch (1986)[18].

$$LS=(\text{flow accumul.}\times\text{cell size}/22.13)^p\ (\sin\alpha/0.0896)^q \quad \textbf{(4)}$$

where p and q are empirical exponents ($p = 0.4$ and $q = 1.3$ [19], flow accumulation signifies the accumulated upslope contributing area for a given cell, cell size is the size of DEM grid cell (for this study is 14.98) and α is the slope degree value.

The C factor reflects the effect of cropping and management practices on erosion rate [6], considering that vegetation reduces the erosive impact of precipitation. This factor ranges between 0 and 1, and is 1 for bare soil. The C factor has a close linkage to land use types. According to the Land Cover Corine 2006 [20] there are three types of soil cover in the study area: 77% of agro-forestry areas, 20% of broad-leaved forest and 3% of wetlands. Vegetation cover can be estimated using vegetation indices derived from satellite images. The most widely used remote-sensing derived indicator of vegetation growth is the Normalized Difference Vegetation Index (NDVI) that ranges from -1 to 1 [21,22]. In this study Landsat TM data was used and the NDVI was therefore computed utilizing band 3 (red) and band 4 (near-infrared) as follows:

$$NDVI=(Band4 - Band3)/(Band4+Band3) \quad \textbf{(5)}$$

In this research satellite images with a spatial resolution of 30 m were used from two different seasons, from February and August of 2007, to compare values. To estimate C factor, the most common procedure using NDVI

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involves the use of regression equation model derived from the correlation analysis between the C factor values measured in the field and a satellite-derived NDVI [21, 23, 24]. Landsat TM images were processed using the IDRISI software [25] and the following formula was used to generate a C factor surface from NDVI values [21]:

$$C= e^{[-\alpha NDVI/(\beta - NDVI)]} \quad \textbf{(6)}$$

where α and β are unitless parameters that determine the shape of the curve relating to NDVI and the C factor. Van der Knijff*et al.* [21] found that this scaling approach gave better results than assuming a linear relationship and the values of 2 and 1 were selected for the parameters α and β, respectively. The C factor map was produced with ArcGIS software.

The support practices factor (P) reflects the effects of specific practices that can be used to reduce the amount and rate of erosion, since the practices modify the flow pattern, grade, or direction of surface runoff [6]. In this case P factor was assigned the value of 1 (no support practice factor), because the support practices in this area are mainly non-existent (before the implementation of the tourism project) and the objective of this first study was to analyze the erosion risk according to the actual conditions.

2.3 Geostatistical analysis

In this study geostatitical tools were used to help to obtain spatial distribution for R and K factor. Geostatistics use stochastical theory of spatial correlation both to predict values at unsampled locations, based on the sampled data, and to assess the uncertainty attached to these predictions [12, 13]. The main tool in geostatistics is the semivariogram, which expresses this dependence [26]. Kriging is one of the most widely used interpolation geostatistical method that assumes that variables close in space tend to be more similar than those further away, minimizing the error variance [27].

The first step for making use of kriging method was to investigate the presence of spatial structure among available data in order to get a better understanding of trends, directional influences and obvious errors. Transformation and trend removal was performed when necessary. Cross-validation was used to compare the prediction performances of the semi-variograms.

3 RESULTS AND DISCUSSION

R-factor values were calculated from over 30 years of rainfall intensity data from all the meteorological stations of south Portugal. The prediction map of the rainfall erosivity was created using ordinary kriging. Erosivity values were found to range in the study region between 477-3603 MJ mm ha^{-1} $h^{-1}yr^{-1}$, for a mean annual rainfall of 515.3 mm. The mean value of the R factor in the study area was 1156 MJ mm ha^{-1} $h^{-1}yr^{-1}$.

Soil erodibility (K) was analyzed according to laboratory results. Descriptive statistics of soil proprieties are given in Table 1. The statistical results of soil properties reflect mostly sandy loam soils that are formed mainly with sand (62.711%), followed by silt (22.013%) and clay (15.276%) with relativity low average of organic matter and moderate to fast hydraulic conductivity (1.5 - 15.1cm/h). Skewness results indicated that almost all properties were normally distributed (skewness between -1 and 1), whereas silt and especially hydraulic conductivity was not normally distributed (skewness more than +1).

Normal distribution is important in order not to produce large prediction errors, however in this case the possi-

TABLE 1 – Descriptive statistics of soil proprieties and RUSLE-K values.

Statistics	Sand (%)	Clay (%)	Silt (%)	Msilt (%)	OM (%)	H.C. (cm/h)	K factor (t ha h $ha^{-1}MJ^{-1}$ mm^{-1})
Mean	62.711	15.276	22.013	35.533	3.631	15.116	0.023
Min	36.470	8.560	5.640	13.710	0.800	1.500	0.003
Max	80.800	27.280	44.980	64.830	7.728	62.400	0.047
Std. Dev.	8.402	3.511	7.764	9.788	1.395	17.552	0.009
Skewness	-0.796	0.623	1.077	0.468	0.630	1.515	0.550
Kurtosis	3.585	3.562	3.638	3.120	3.649	3.941	3.601

TABLE 2 –Cross-validation results of the fitted semi-variogram models used to create the prediction maps of soil properties and K factor values.

Soil property	Sand (%)	Clay (%)	Silt (%)	MSilt (%)	OM (%)	Hydr C. (cm/hr)	K factor
Model	Pentaspherical	Pentaspherical	Exponential	Exponential	Exponential	Exponential	Exponential
Nugget	44.567	4.7433	42.606	68.544	0.89201	0	0.00005551
Sill	17.534	8.9499	15.558	50.649	1.1115	10.381	0.000035367
Range	921.785	640.178	3323.6	5104.63	277.8	372.49	5104.63
ME	0.07544	-0.01481	-0.03616	-0.04387	-0.00072	-0.00283	-0.000032
RMSE	8.162	3.352	7.687	9.012	1.384	3.161	0.00799
ASE	7.458	3.358	6.978	9.093	1.402	3.205	0.00812
RMSSE	1.093	0.9883	1.099	0.9949	0.9891	0.9829	0.9893
Nugget/ Sill	2.541747	0.529984	2.738527	1.353314	0.802528	0	1.569542229

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bility of error is low (see Table 2). Table 2 summarily presents the indicators which facilitated the choice of the most appropriate model of semivariogram for the creation of prediction maps.

The cross-validation results show that the mean values of error (ME) were very low, i.e. close to zero, and the root mean square standardized of error (RMSSE) is close to 1 which shows that the estimation had an acceptable accuracy and the prediction values are close to the measured values. The root mean square (RMSE) and average standardized error (ASE) are indices that represent the quality of prediction and should be low. The nugget-to-sill ratio presented in Table 2 indicted weak spatial dependence for almost all soil properties, which reflects high variance at short distances (high heterogeneity). Moderate and high spatial dependence was obtained for clay and hydraulic conductivity, respectively. Cambardella et al. [28] suggested that strong spatial dependency existed if the nugget semi-variance of a variable expressed as a fraction of the total semi-variance was <0.25, moderately spatially dependent if it was 0.25 to 0.75, and weakly spatially dependent if the ratio was >0.75. According the same author, weak spatial dependence is controlled by non-intrinsic changes such as inappropriate management.

The prediction map of soil erodibility (K), obtained through nomograph previously presented, is presented in Figure 2.

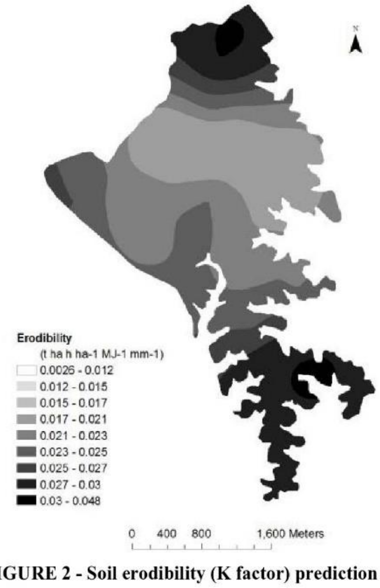

FIGURE 2 - Soil erodibility (K factor) prediction map.

The K factor values were predicted to vary from 0.0026 to 0.047 t ha h ha^{-1} MJ^{-1} mm^{-1}, with a mean value of 0.023 t ha h ha^{-1} MJ^{-1} mm^{-1}. From the map derived it could be seen that the highest soil erodibility K values are mainly located in the south and north sections, where the highest amount of susceptible particles (silt and very fine sand) are found. These areas will require special attention towards land-use changes (tourism development) because these are suspected to have more susceptibility to erosion. Soils with high permeability are more resistant to erodibility, and this was confirmed with the results.

Figure 3 shows the typical NDVI values in the winter (February of 2007) and in the summer (August of 2007), and the corresponding estimated C factor. NDVI values in the winter in this area are higher with a maximum of 0.714 and an average of 0.434. In the summer the highest value is 0.274 with an average value of 0.102. It shows that the vegetation in wet period (winter) is denser than in dry period (summer), as is expected. Availability of water in the soil and moderate temperatures in winter facilitate natural vegetation growth.

By analyzing the maps it can be concluded that C Factor has a negative correlation with NDVI values and that the highest values of NDVI caused the lowest values of C factor, resulting in lowest erosion values. The results revealed that C factor values in winter are lower than in summer.

Regarding LS factor it was found that the elevation in this area, according to the DEM produced, ranges between 145-215 meters. The slope percentages map created is shown in Figure 4. Slope values in this study area vary from 0 to 36% with an average of 5.4%, whereas only 2.5% of total area exceeds a slope of 15%. The highest slopes result in an increased overland flow, rilling and concentrated flow depth. LS factor in the study area, which depends on slopes and flow accumulation, varies from 0 to 28.96, with mean and standard deviation of 1.62 and 1.81, respectively

The results demonstrate that LS factor has a clear correlation with slope, because areas with highest slope values have the highest LS factor values. The highest values of LS occur in the centre of the area, in the southwest part and the east side close to the dam.

The RUSLE factors were integrated within the raster calculator option of the ArcGIS spatial analyst to obtain and quantify soil erosion rates using RUSLE equation. The spatial distribution of soil erosion is shown in Figure 5. These prediction maps are created according to the different C factor values from the winter and summer of 2007. As can be seen from the maps, the risk of soil erosion is higher in the summer in respect to the average annual erosivity. According to C factor estimated in winter, the average soil erosion is 14.1 t $ha^{-1}year^{-1}$. Taking into account the vegetation cover in summer, the predicted average value is 38.1 t $ha^{-1}year^{-1}$. The maps show that the most sensitive areas are located in the southwest and east (north and central) part. The distribution of soil erosion indicates that severely eroded areas mainly lie in higher slopes. In this area soil erosion is highly dependent on the local terrain and land-use.

There are significant differences between winter and summer seasons, which are very noticeable in Figure 6.

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Erosion drastically increases in summer, where risk of less than 8 t $ha^{-1}year^{-1}$decreases from 52% of the area in winter to just 22% in summer. Furthermore, erosion with sedimentation between 30-45 t $ha^{-1}year^{-1}$increases from 6% to 17% and erosion with more than 45 t $ha^{-1}year^{-1}$ sedimentation increases from 5.1% to 31% in summer. Erosion over 30 t $ha^{-1}year^{-1}$during summer season constitutes almost half of the area (48%) whereas the same size of the area

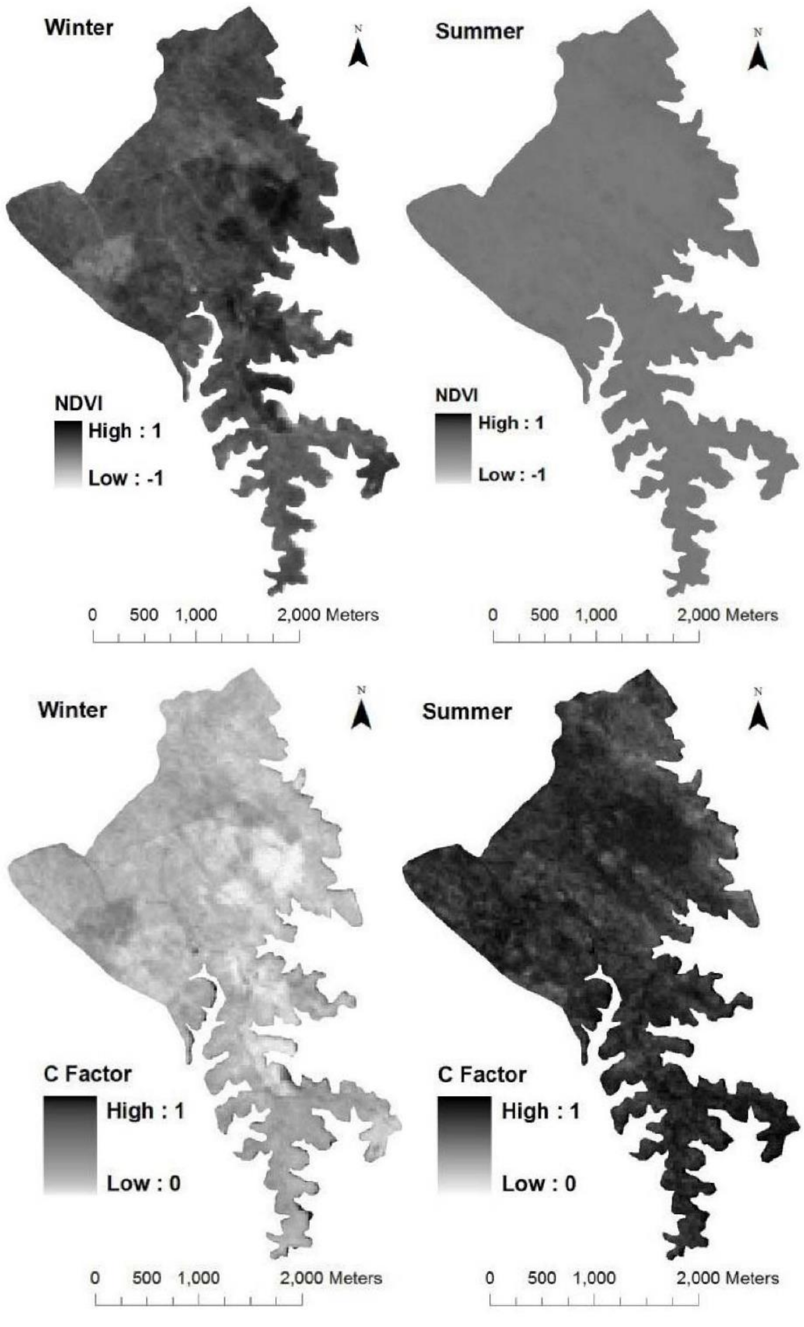

FIGURE 3–NDVI and cover management factor maps (C-factor).

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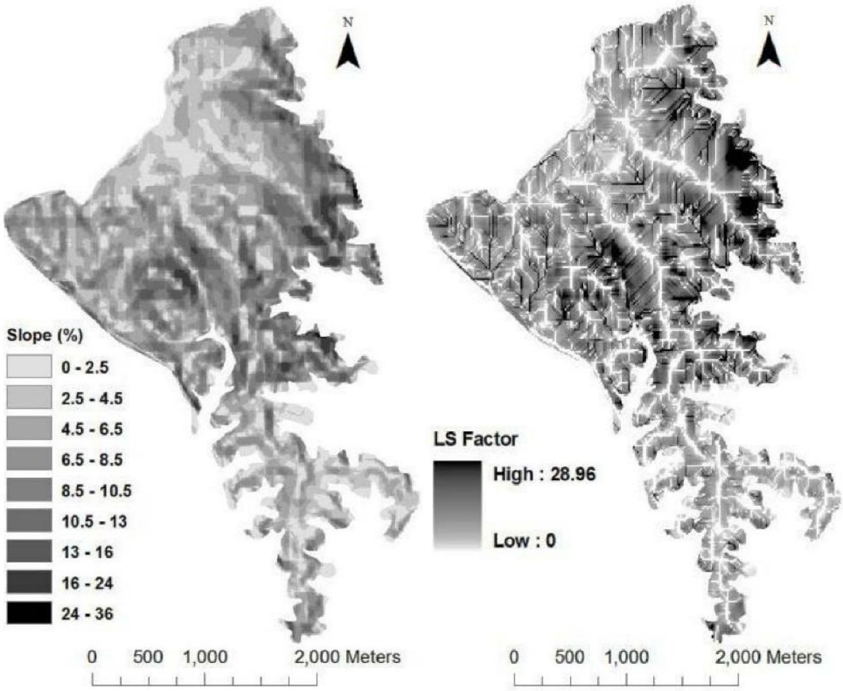

FIGURE 4 - Slope percentages map and LS map of the study area.

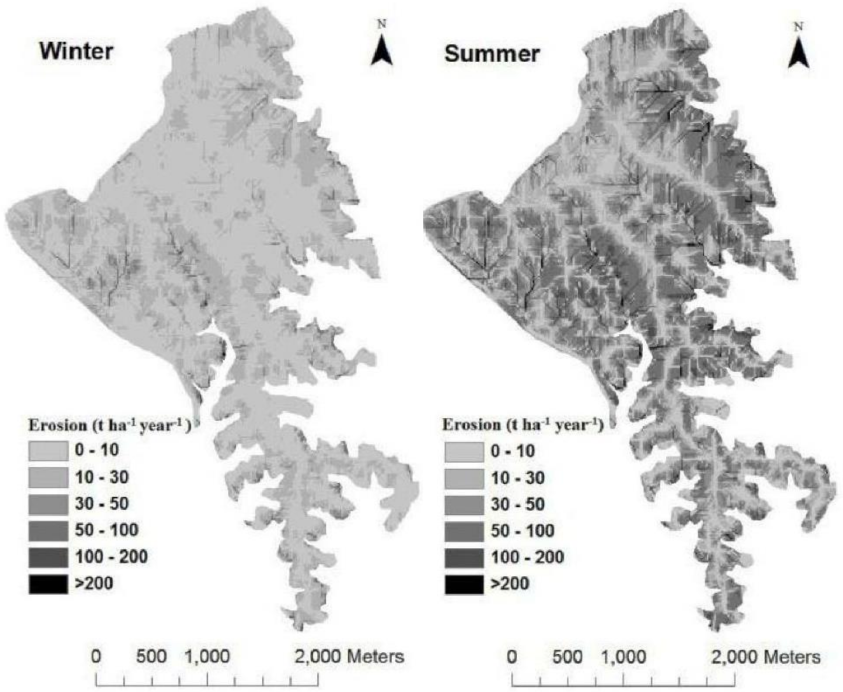

FIGURE 5 – Soil erosion predicted maps according differences in vegetation in winter and summer seasons.

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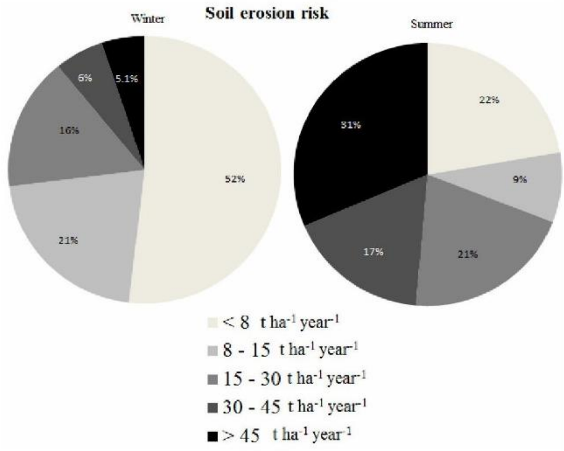

FIGURE 6 – Variances in soil erosion risk for different seasons.

has erosion over 8 $tha^{-1}year^{-1}$in winter. This clearly demonstrates the large impact of vegetation cover on potential soil erosion.

The results of the application of RUSLE model are consistent with those obtained for other Mediterranean watersheds with similar characteristics [21, 22].

The study area has not been cultivated for years and during summer the vegetation cover was low and vulnerable to erosive rains. Usually the months with higher average monthly rainfall coincide with periods of maximum vegetation vigour, and the months of lower rainfall (summer) with the lowest vegetation cover rates. However, the most erosive events occur at late summer and early autumn, when rain storms start to occur. In that period vegetation cover is at its minimum and soils are very susceptible to erosion after the long dry season. In agriculture areas with typical winter cultures, autumn is a significantly critical period for soil erosion because the land is prepared for seeding and doesn't have vegetation cover due to tillage. Furthermore climate change could increase the magnitude of these phenomenons.

These results show that it is important to maintain the soil cover all through the year, especially in the higher precipitation period, to avoid high soil erosion rates. Ongoing human intervention, ever-changing cultivation and forest degradation in the watersheds could increase sediment yield in the future. It should be emphasized that the areas with high susceptible soil condition would need special priority for implementation of soil conservation practices. For effective watershed planning there must be a close coordination between vegetative and structural control measures. This can be done by using annual cultures.

The evaluation of soil erosion probability zones and soil loss at the watershed scale is essential for comprehensive local and regional development and sustainability. The land-use of the study area changed significantly through tourism development projects related to golf areas and other rural recreation developments. This will provide vegetation cover during all year, which means more protection for soil and a lower C factor. Since those soils were subjected to intensive agriculture in the past and were abandoned in the last years, changes in vegetation management due to land-use change are positive, increasing organic matter, improving permeability and decreasing soil erodibility (K factor). However, in the long term, the introduction of recreational areas may increase soil compaction, decreasing soil infiltration and increasing runoff [29]. Therefore it is important to be aware on adequate soil management and conservation solutions (P factor) to any land use change at the watershed.

Although RUSLE approach does not differentiate between seasonal changes, there is evidence in literature that suggests soil erosion may change according to intra-annual rainfall figures and vegetation phenology. A study by Panagos *et al.* [30] was done to evaluate changes in erosion risk rates depending on season. Their results demonstrated the importance of taking seasonal variations of Mediterranean climate into account. This study also confirmed the importance of seasonal erosion mapping to identify the critical periods. Understanding seasonal variations is essential to delineate appropriate strategies. Problems associated with the highest erosion, for the period when the vegetation cover is low, and the possibility of climate changes, could be solved by restricting soil-conservation management techniques.

4 CONCLUSIONS

This study confirmed that the combination of GIS and erosion yield models is a successful procedure to determine the spatial distribution of soil erosion risk and sediment yield under a variety of simulation scenarios. This study also demonstrated that geostatistical techniques can be useful to estimate soil erosion and its factors at unsampled locations, based on the sampled data. The prediction maps provide useful information for planning the future land-use changes and to apply soil conservation practices at the identified high-risk areas. Those results are essential to increase the knowledge about local conditions and to create a solid base for a Decision Support System that will provide site specific measures to protect soil as well as to decrease the risk of soil erosion, reduce costs, increase the dam's life span and promote sustainability of these ecosystems.

At this point our study has evaluated the effect of seasonally changing vegetation cover on potential erosion risk. It was confirmed the importance of seasonal erosion mapping to identify the critical periods. It is important to continue assessment with intra-annual rainfall variation impact. This can further facilitate the establishment of a new RUSLE modelling approach. This new approach would allow erosion figures to be linked to specific land uses and therefore develop adequate measures.

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Appendix III

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Seasonality of Soil Erosion Under Mediterranean Conditions at the Alqueva Dam Watershed

Vera Ferreira · Thomas Panagopoulos

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Abstract The Alqueva reservoir created the largest artificial lake of Western Europe in 2010. Since then, the region has faced challenges due to land-use changes that may increase the risk of erosion and shorten the lifetime of the reservoir, increasing the need to promote land management sustainability. This paper investigates the aspect of seasonality of soil erosion using a comprehensive methodology that integrates the Revised Universal Soil Loss Equation (RUSLE) approach, geographic information systems, geostatistics, and remote-sensing. An experimental agro-silvo pastoral area (typical land-use) was used for the RUSLE factors update. The study confirmed the effect of seasonality on soil erosion rates under Mediterranean conditions. The highest rainfall erosivity values occurred during the autumn season (433.6 MJ mm ha^{-1} h^{-1}), when vegetation cover is reduced after the long dry season. As a result, the autumn season showed the highest predicted erosion (9.9 t ha^{-1}), contributing 65 % of the total annual erosion. The predicted soil erosion for winter was low (1.1 t ha^{-1}) despite the high rainfall erosivity during that season (196.6 MJ mm ha^{-1} h^{-1}). The predicted annual soil loss was 15.1 t ha^{-1}, and the sediment amount delivery was 4,314 × 10^{3} kg. Knowledge of seasonal variation would be essential to outline sustainable land management practices. This model will be integrated with World Overview of Conservation Approaches and Technologies methods to support decision-making in that watershed, and it will involve collaboration with both local people and governmental institutions.

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Keywords RUSLE · Seasonality · Soil erosion · Sustainable land management (SLM)

Introduction

Soil erosion is one of the most dynamic and complex environmental and economic issues in the Mediterranean region, related to land-use changes and possibly climate changes (López-Bermudez et al. 1998; Yang et al. 2003; Nearing et al. 2004). Over the past few decades, the combination of natural and anthropogenic factors has induced an increase in soil erosion risks (Kosmas et al. 1999; Blavet et al. 2009). Soil erosion, which has become a threat to sustainability, is characterized by the decline of soil depth and quality in one place and deposition in another, due to the increase of runoff (Lal 2001; Prasannakumar et al. 2011). Therefore, it often causes negative downstream impacts, such as the sedimentation in rivers and reservoirs that reduces their storage capacity and life span (Pandey et al. 2007).

The semiarid Mediterranean regions are particularly vulnerable because they have a long dry period followed by heavy bursts of intensive rainfall, falling on steep slopes with fragile soils and sparse vegetation (Karydas et al. 2009). Furthermore, these Mediterranean regions have been subject to substantial land-use changes over the past century due to changes in agricultural technologies and social, political, and economic development (Bakker et al. 2008). Some investigations (Kosmas et al. 1997; Wang et al. 2003; Long et al. 2006; Cerdan et al. 2010; Cantón et al. 2011) have confirmed that there is a strong connection between land-use change and soil erosion because they found that these changes in vegetation and farming techniques greatly affected the intensity and frequency of overland flow and surface wash erosion. According to the CORINE program, Mediterranean countries, such as Portugal and Spain, face the greatest risks of erosion (Desir and Marín 2007), and areas at high risk of erosion in Portugal cover almost one-third of the country (Grimm et al. 2001).

Monitoring soil erosion in situ is very costly and usually limited to small experimental sites. Several initiatives have been made, during the last decades, to assess soil erosion, and there has been significant research into appropriate sediment yield models (Merrit et al. 2003; Bhattarai and Dutta 2008), namely physical models (e.g., Pan-European Soil Erosion Risk Assessment, Kirkby et al. 2004), conceptual models (e.g., Soil and Water Integrated Model, Krysanova et al. 1998), and empirical models (e.g., Universal Soil Loss Equation, Wischmeier and Smith 1978). These models differ in terms of complexity, processes considered and the data required, so there is no "best" model for all applications because they depend on the intended use and the characteristics of the catchment considered (Volk et al. 2010). The empirical Universal Soil Loss Equation (USLE) is one of the most widely used models for estimating annual soil loss (Wischmeier and Smith 1978), from agricultural watersheds, and its modifications include the Revised Universal Soil Loss Equation (RUSLE) (Renard et al. 1997). Significant USLE disadvantages consist of the overestimation of erosive slope lengths and the lack of possibilities for process-oriented simulations like sediment transport (Winchell et al. 2008). However, the RUSLE is easy and beneficial to use, compared with most other models, because of its structural simplicity, fewer input data requirements and the availability of parameter values (Volk et al. 2010).

The USLE-family models are normally applied to assess the annual soil loss and do not consider seasonal changes, despite the evidence in the literature which confirms that soil erosion changes rapidly during the year according to intra-annual rainfall rates and vegetation cover (Van Leeuwen and Sammons 2003; Lu et al. 2003; Grazhdani and Shumka 2007; Panagos et al. 2011). Many soil erosion modeling efforts often incorporate static maps of vegetation classes, climate, relief and soils, omitting seasonal patterns of climate, and vegetation growth (Van Leeuwen and Sammons 2003).

There are some RUSLE factors that can be considered time-invariant (not- or little-changing over time), such as topography (LS factors) and most of the management measures (e.g., terracing) (P factor). On the other hand, rainfall erosivity (R factor) and vegetation cover (C factor) are highly time-variant (changing substantially over time) and have been assessed for each season (Van der Knijff et al. 1999; Van Leeuwen and Sammons 2003; Panagos et al. 2011; Alexandridis et al. 2013). Soil erodibility (K factor) displays an intermediate state because it is influenced by intrinsic properties that change very slowly (e.g., soil texture) and dynamic properties that change more quickly (e.g., soil crust) (Song et al. 2005). However, it is hard to evaluate the seasonal variation of soil erodibility for large geographic scales; thus, it is usually considered constant (Panagos et al. 2011; Alexandridis et al. 2013).

Rainfall erosivity depends on the climate area, the seasonal pattern of rainfall and the random occurrence of storms (Evrard et al. 2010). Rainfall under a Mediterranean climate is characterized by high temporal variability not only over the years but also throughout the year (González-Hidalgo et al. 2007). Several studies, many of them from Mediterranean regions, have taken into account the annual variation of precipitation that results in a strong seasonality of erosion rates (Renschler et al. 1999; Cerdá 2002; Regüés and Gallart 2004; Lana-Renault et al. 2007; Boardman et al. 2009; Diodato and Bellocchi 2010).

Vegetation cover is an important factor to protect soil against erosion, and the efficiency varies greatly with vegetation types, which are always related to land-use patterns (Zhou et al. 2006). This RUSLE factor is frequently derived from satellite images; however, because of the lack of data, very few images (or only one image) are used per year, despite being temporally variable according to plant phenology (Alexandridis et al. 2013). Vegetation cover usually presents temporal dynamics due to differences in availability of water in the soil and temperatures, and it is often induced by management practices. Some investigations have focused on these seasonal changes of vegetation cover (López-Bermudez et al. 1998; Camacho-De Coca et al. 2004; Gallo et al. 2005).

The seasonality approach on soil erosion processes can be essential for successful conservation and sustainable land management (SLM) in the Mediterranean regions, primarily to address specific management practices throughout the year in the different land-use scenarios. For an adequate SLM strategy, besides the scientific knowledge (e.g., prediction maps on soil erosion), it is significant the collaboration with projects as World Overview of Conservation Approaches and Technologies (WOCAT). The WOCAT is a global network that aims to promote improved decision-making on land management, sharing valuable knowledge with specialists and practitioners from all over the world (WOCAT 2007; Liniger and Schwilch 2002), allowing the knowledge exchange WOCAT provide tools to identify situations in need of action and an SLM database with a full range of different case studies documented from all over the world (Schwilch et al. 2012). The WOCAT tools were integrated with a stakeholder learning approach and a decision-support system for identifying, documenting, and selecting SLM strategies on the EU-DESIRE project (Schwilch et al. 2012). The EU-DESIRE research project provided an excellent opportunity to develop and test a generic decision-support system on land-degradation mitigation.

Relevance and Objectives

Management of reservoirs is of major importance regarding the water supply in Portugal. The Alqueva reservoir constitutes the largest artificial lake in Western Europe; however, its capacity cannot be maintained due to an annual deposition of sediment. The region surrounding the Alqueva reservoir has new challenges, as traditional land-uses and human activities are changing, and new risks are arising. The possible land-use changes in this region will be due to tourism development (construction of golf courses, hotels, and marinas), intensification of irrigated farming (vineyards, olive orchards, maize, etc.) and, consequently, the abandonment of the extensive agro-silvo pastoral system (the actual land-use studied and described in the following section).

Land-use changes in the study area are increasing the need to promote the sustainability of the ecosystems to decrease soil erosion, augmenting the reservoir's life span. The present study is part of a broader project with a major aim to investigate soil erosion under different land-use scenarios, looking forward to more SLM through the application of a decision-support system (DSS) in the Alqueva watershed. Therefore, the main objective of the present research is to improve understanding of the seasonality of soil erosion, under Mediterranean climatic conditions, at a case study area of the Alqueva region, due to intra-annual variations in rainfall and vegetation cover. We used the RUSLE equation in combination with geographic information systems (GIS), geostatistics, and remote-sensing to estimate annual and seasonal soil erosion from a typical land-use of the Alqueva region.

Study Area

The Alqueva reservoir is located on the river Guadiana in Alentejo, a semiarid region in the south of Portugal (7°30′W, 38°15′N). In 2010, the lake was filled to the planned level, with a surface area of 250 km^2 (from which 35 km^2 is in Spain) and a total capacity of 4.15 m^3. The total shoreline of the lake is approximately 1,100 km, extends for 83 km in length (Lindim et al. 2011). The complex project was constructed during 1998–2002, and the main objective was to create a strategic water reserve, for supply water to the populations; irrigation for farms in the surrounding area (about 110,000 ha), to produce hydroelectric power; and a large reservoir where several tourist projects are also being built. The Alqueva reservoir has a direct influence in the regions surrounding it (namely 18 counties).

This region of Alentejo is characterized by a highly heterogeneous and complex landscape structure. A typical landscape is named "Montado," an agro-silvo pastoral system in which agricultural, forest, and pastoral activities complement each other (Borges et al. 2010). The traditional "Montado" is comprised of an open formation of cork oak (*Quercus suber*) and holm oak (*Quercus ilex*), in varying densities, combined with a rotation of crops/fallow/pastures. In some montado areas, oaks are mixed with olive trees. In this region, in the beginning of the twentieth century, there was an intensification of agriculture, and the area was used especially for cereal production in combination with extensive livestock breeding. This intensification led to numerous environmental impacts particularly increased soil erosion. However, especially since Portugal joined the European Community in 1986, the abandonment of agricultural activities in Alentejo has increased, and the agro-silvo pastoral system is in transition toward a silvo-pastoral or even purely forestry system (Pinto-Correia and Mascarenhas 1999). This change left the ecosystems more vulnerable to fires that increased its susceptibility to soil erosion (Jones et al. 2011); however, in some areas it resulted in the decrease of soil erosion (Bakker et al. 2008). Today, around the Alqueva reservoir, the intensification of some farming systems occurs simultaneously with the reduction and ultimate abandonment of others (Jones et al. 2011).

The climate is Mediterranean with very hot and dry summers and mild winters, as well as with higher rainfall during the autumn season. The annual average temperature ranges from 24 to 28 °C in hot months (July/August) and from 8 to 11 °C in cold months (December/January). The average annual precipitation at the nearest meteorological station, for the last 30 years, is 498.9 mm. The region is affected by intense dry periods without precipitation, since 80 % of the precipitation occurs from October to April.

An experimental research site (Fig. 1) was primarily chosen to study the potential risk of soil erosion and to better understand seasonal variations under Mediterranean conditions. Therefore, a typical land-use was studied from October of 2006 to September of 2009. During this time, the land-use in the area was a "montado"—an agro-silvo pastoral system in transition toward a silvo-pastoral. Since 2004 this area has not been cultivated, and the soil had been kept as pasture with natural vegetation. The experimental area is 782 ha and lies in Herdade do Roncão (Roncao D'el Rey) beside the reservoir in the west part, near the Regengos de Monsaraz. Currently, there is a

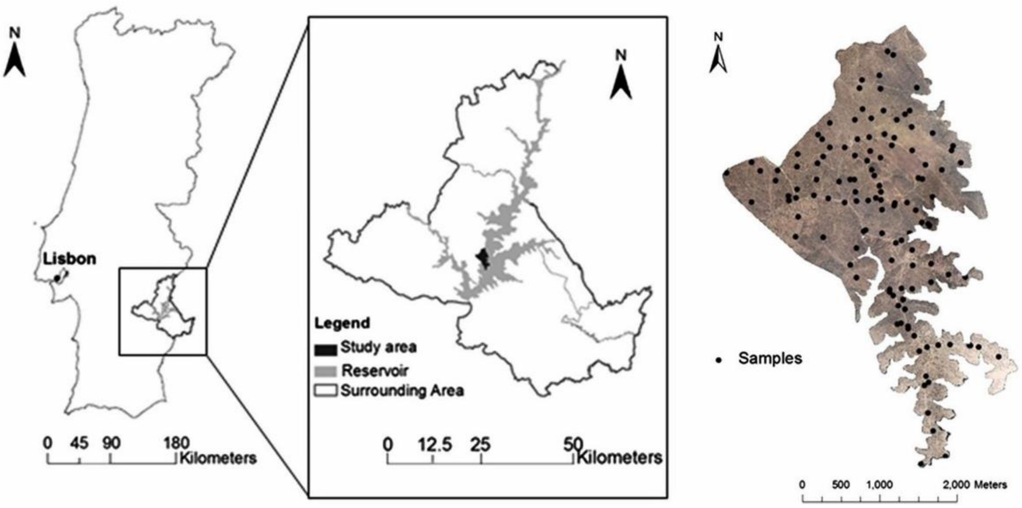

Fig. 1 Location of the Alqueva reservoir and the experimental study area (Roncão)

tourism project that will be implemented at this site (included in the "Parque do Alqueva" project), taking advantage of the attractive landscape and water availability.

Materials and Methods

The development of soil loss models has simplified the study of soil erosion, especially for conservation purposes, to provide an effective and acceptable level of accuracy. Different projects at large scales have been created with the contribution of the European Commission, in which soil erosion risk was estimated for different European countries including Portugal (e.g., PESERA, Kirkby et al. 2004) and vulnerable areas that were identified for the Mértola municipality (Alentejo) (e.g., MEDALUS, Kosmas et al. 1999). Furthermore, real soil loss was measured for a period of 44 years in field-experimental plots with approximately 166.7 m^2 ha, including in the Alentejo area, at the Vale Formoso experimental center (Roxo and Cortesão Casimiro 2004). Nevertheless, as already mentioned, there is a need to quantify soil erosion under different, typical land-uses scenarios, in the Alqueva watershed, with high resolution and looking toward SLM. Therefore, we selected a RUSLE approach because of its flexibility and simplicity, as well as the fewer input data requirements. RUSLE is defined as:

$$A = RKLSCP, \tag{1}$$

where A is the potential erosion (computed annual average soil loss in t ha^{-1} $year^{-1}$), R the rainfall and runoff erosivity factor, K the soil erodibility factor, LS is the slope length and gradient factor, C is the vegetation cover factor, and P is the support practice factor RUSLE (Renard et al. 1997). In this research, some adjustments in the RUSLE model were necessary to account for the effect of seasonality in rainfall and vegetation data. So, to study seasonal soil erosion, the rainfall erosivity (R) and vegetation cover (C) factors were analyzed per season, and the topographic (LS) factor, the protection measures (P), and erodibility (K) were aggregated as "static." We consider soil erodibility (K) as constant since the soil was not disturbed in this land-use during the study time. According to Bryan (2000), under undisturbed conditions they reach an equilibrium level which remains fairly constant at any location.

GIS techniques, geostatistical methods, and remote-sensing have been successfully used in combination with these models because they make soil erosion estimation and its spatial distribution feasible, with reasonable costs and accuracy (Bhattarai and Dutta 2007; Terranova et al. 2009). Remote-sensing allows land-cover change detection (Wang et al. 2009). Geostatistical tools have been used to produce prediction maps, allowing the estimation of numerical and categorical attribute values in unsampled locations (Goovaerts 1999a). Soil experimental data, remote-sensing techniques, land-use inventory, digital elevation data, and rainfall erosivity data were used as resource data sets to estimate seasonal soil erosion. The spatial data was treated with ArcGIS software (ESRI 2008).

Soil Erodibility Factor (K)

Soil erodibility factor (K) represents the susceptibility of a soil to be eroded, and it is defined as the average rate of soil loss per unit of rainfall erosivity index from a cultivated continuous fallow plot, on a 9 % slope 22.1 m long

(Renard et al. 1997). The soil erodibility factor (K) is a quantitative value, experimentally determined, based on USLE literature, using an algebraic approximation (Wischmeier and Smith 1978):

$$K = [2.1 \times 10^{-4}(12 - \mathrm{OM}) \times \mathrm{M}^{1.14} + 3.25\,(s-2) + 2.5(p-3)]/100, \quad (2)$$

where OM is organic matter, s is soil structure, and p is permeability class. M is the product of the primary particle size fractions (% MSilt) × (%MSilt + %MSand), where %MSilt is percent modified silt (0.002–0.1 mm), and %MSand is percent modified sand (0.1–2 mm). Modified silt is the amount of silt particles and very fine sand, considered the most susceptible particles to erosion because they can be easily removed by the raindrop splash and runoff water.

To evaluate this factor, a total of eighty-two (82) soil samples of about 1 kg from 0 to 20 cm depth were collected in the experimental area. In the laboratory, the individual samples were analyzed: soil texture, using the Bouyoucos hydrometer method (Bouyoucos 1962), and OM, using the Walkley and Black (1934) method. To estimate the permeability, the field-saturated hydraulic conductivity was measured in the field using a Turf-Tec double-ring infiltrometer. Permeability class and soil structure class were defined in accordance with Renard et al. (1997).

Computed K factor and soil properties for each soil sample unit were added into ArcGIS software, statistics were studied and a continuous surface representing the spatial distribution was prepared using a Geostatistical Analyst tool. The interpolation method used for raster creation was ordinary kriging (OK), which is a common method for data interpolation that gives the most accurate results (Goovaerts 1999a). The first step for making use of the OK method was to investigate the presence of spatial structure among available data in order to get a better understanding of trends, directional influences and obvious errors. When necessary, trend removal was performed and transformation methods (Box–Cox, Arcsine, and Log) were applied for not normally distributed data. Semivariograms were produced for each property and the cross validation was used to find which semivariogram model gives the most accurate predictions for map production. Cross validation compares predicted and mentioned values through calculated statistics (mean error [ME], root-mean-square [RMSE], average standard error [ASE], and root-mean-square standardized error [RMSSE]). The ME can be negative or positive because we can have an over-prediction or under-prediction at any point. Closer values of the ME to zero and closer values of the RMSE to 1 signified that the prediction values were close to measured values (Wackernagel 1995). Where models presented similar values for ME and RMSSE, the lowest values of RMSE and ASE were taken into consideration. Additional model parameters, such as nugget and sill, helped to choose the most appropriate model of the prediction maps (Isaaks and Srivastava 1989).

Slope Length and Steepness Factor (LS)

According to USLE literature, slope length (L) is defined as the horizontal distance from the origin of overland flow to the point where either the slope gradient decreases enough that deposition begins or runoff becomes concentrated in a channel. The slope steepness (S) shows the influence of slope gradient on erosion (Wischmeier and Smith 1978). Direct measurements of slope and slope length were initially proposed to evaluate these factors (Renard et al. 1997). However, this method is only suitable for small plots and parcels because intensive field measurements are obviously not feasible on a regional scale. At the watershed scale, the use of a digital elevation model (DEM) in GIS, for data input, is a better approach (Nekhay et al. 2009).

In this study, to estimate this factor we created a DEM in ArcGIS software using The Shuttle Radar Topography Mission (SRTM) data of 30 m resolution. GIS analyses allow users to generate slope steepness (S) and slope length (L) raster covers by a number of different methods. In this case, the combined LS factor (without units) was computed for the watershed by means of the ArcGIS spatial analyst extension using DEM, following Eq. (3), as proposed by Moore and Burch (1986)

$$\mathrm{LS} = (\text{flow accumulation} \times \text{cell size}/22.13)^{\mathrm{p}}\,(\sin\alpha/0.0896)^{\mathrm{q}}, \quad (3)$$

where p and q are empirical exponents ($p = 0.4$ and $q = 1.3$) (Moore and Wilson 1992), flow accumulation signifies the accumulated upslope contributing area for a given cell, cell size is the size of the DEM grid cell (for this study, it is 30 m) and α is the slope degree value.

Rainfall Erosivity (R)

The rainfall–runoff erosivity (R) is usually known as one of the most important indicators for erosive potential of raindrops impact. It also reflects the potential of runoff generated by erosive storms. According to Renard et al. (1997), the rainfall–runoff factor is determined through the sum of erosive storm values EI30 occurring during a mean year, which is the product of total storm kinetic energy (E) times the maximum 30 min intensity (I30), where E is in MJ ha^{-1} and I30 is in mm h^{-1}. When detailed information about rainfall every 15 or 30 min is not available, the EI30 index can be estimated from daily and monthly values of precipitation. In this study, we estimated the monthly erosive storm empirical index (EI30) using the regression equation calibrated by Goovaerts (1999b), with high correlation ($r^2 = 0.92$):

$$EI30_{month} = 6.56 \times rain_{10} - 75.09 \times days_{10}, \quad (4)$$

where $rain_{10}$ is the monthly rainfall for days where precipitation exceeds 10 mm, and $days_{10}$ is the monthly number of days where precipitation exceeds 10 mm. This equation was calibrated from measurements recorded at 17 tipping-bucket raingauges (time resolution = 1 min) in the south of Portugal, between October 1992 and March 1995. Through the equation, the monthly EI30 values were estimated for 25 meteorological stations in the surrounding area of the Alqueva, using daily precipitation values during 30 years (1979–2009), to accommodate apparent cyclical rainfall patterns. A period of 20–25 years is recommended for computing the average R (Wischmeier and Smith 1978). We used this data over different seasons, considering autumn (October to December), winter (January to March), spring (April to June), and summer (July to September). Rainfall erosivity maps, for the surrounding area, were created for each season also using geostatistic techniques (OK) (see "Soil Erodability Factor (K)" section).

Vegetation Cover Factor (C)

According to USLE literature, the C factor reflects the effect of vegetation on erosion rate (Renard et al. 1997). Vegetation cover protects the soil by dissipating the raindrop energy before it reaches the soil surface. As such, soil erosion can be effectively limited with proper management of vegetation, plant residue, and tillage (Lee 2004). This factor ranges between 0 and 1; it is 1 for bare soil, and it has a close linkage to land-use types.

The C factor has been one of the most difficult RUSLE coefficients to estimate because it is derived based on empirical equations with measurements of ground cover, aerial cover, and minimum drip height (Wischmeier and Smith 1978). Several methodologies have been developed for estimating vegetation cover as an alternative to C values derived from USLE tables. Remote-sensing has been one of the most widely used methods for mapping the C factor (De Jong 1994; Kouli et al. 2009; Prasannakumar et al. 2011) because vegetation cover can be estimated using vegetation indices derived from satellite images such as Normalized Difference Vegetation Index (NDVI). NDVI is an indicator of vegetation growth and ranges from -1 to 1 (Van der Knijff et al. 1999). This method gives a different perspective on soil erosion studies because it allows the estimation of intra-annual changes in vegetation cover through images for different periods (Ouyang et al. 2010).

In this study, Landsat TM data from October 2006 to September of 2009 was used. We took one satellite image from each season due to the difficulty in getting data for each month (a total of 12 satellite images with 30 m resolution).

NDVI was therefore computed utilizing band 3 (red) and band 4 (near-infrared) as follows:

$$NDVI = (Band4 - Band3)/(Band4 + Band3), \quad (5)$$

To estimate the C factor, the most common way of using NDVI involves the use of a regression equation model derived from the correlation analysis between the C factor values, measured in the field using the line-point intercept sampling method (Herrick et al. 2005), and a satellite-derived NDVI (De Asis and Omasa 2007). Landsat TM images were processed using the IDRISI software (Eastman 2006), and the following formula was used to generate a C factor surface from NDVI values (Van der Knijff et al. 1999):

$$C = \exp(-\alpha(NDVI/(\beta - NDVI)), \quad (6)$$

where α and β are unitless parameters that determine the shape of the curve relating to NDVI and the C factor. Van der Knijff et al. (1999) found that this scaling approach gave better results than assuming a linear relationship, and the values of 2 and 1 were selected for the parameters α and β, respectively. C factor maps were produced for different seasons with ArcGIS software.

Conservation Practice Factor (P)

The support practices factor (P) reflects the effects of specific practices that can be used to reduce the amount and rate of erosion, such as contouring, strip-cropping, terracing, and subsurface drainage. These practices affect erosion by modifying the flow pattern, grade, or direction of surface runoff and by reducing the amount and rate of runoff (Renard et al. 1997). In this case, the P factor was assigned the value of 1 (no support practice factor) because the support practices were nonexistent for actual conditions in this area (agro-silvo pastoral system in transition toward a silvo-pastoral or even purely forestry system).

Results and Discussion

Soil Erodibility (K)

Soil erodibility (K) and specific soil proprieties were analyzed for each sample. The descriptive statistics and cross validation indicators are given in Table 1. As can be seen in the table, all semivariogram models provide low mean error (ME), i.e., close to zero, and the root mean square

Table 1 Descriptive statistics and cross validation results of semivariogram models used to create the prediction maps of soil properties and K factor values

Statistics	Sand (%)	Clay (%)	Silt (%)	Msilt (%)	OM (%)	HC (cm h^{-1})	K factor (t ha h ha^{-1} MJ^{-1} mm^{-1})
Mean	62.711	15.276	22.013	35.533	3.631	15.116	0.023
Min	36.470	8.560	5.640	13.710	0.800	1.500	0.003
Max	80.800	27.280	44.980	64.830	7.728	62.400	0.047
SD	8.402	3.511	7.764	9.788	1.395	17.552	0.009
Cross validation	**Sand (%)**	**Clay (%)**	**Silt (%)**	**Msilt (%)**	**OM (%)**	**HC (cm h^{-1})**	**K factor (t ha h ha^{-1} MJ^{-1} mm^{-1})**
Model	Pentaspherical	Pentaspherical	Exponencial	Exponencial	Exponencial	Exponencial	Exponencial
ME	0.07544	−0.01481	−0.03616	−0.04387	−0.00072	−0.00283	−0.000032
RMSE	8.162	3.352	7.687	9.012	1.384	3.161	0.00799
ASE	7.458	3.358	6.978	9.093	1.402	3.205	0.00812
RMSSE	1.093	0.9883	1.099	0.9949	0.9891	0.9829	0.9893
Nugget/sill	2.541747	0.529984	2.738527	1.353314	0.802528	0	1.569542229

Min minimum, *Max* maximum, *SD* standard deviation, *Msilt* modified silt (0.002–0.1 mm), *OM* organic matter, *HC* hydraulic conductivity, *K* soil erodibility, *ME* mean error, *RMSE* root-mean-square, *ASE* average standard error, *RMSSE* root-mean-square standardized error

standardized of error (RMSSE) is close to 1, which shows that the estimation had an acceptable level of accuracy. The root mean square (RMSE) and average standardized error (ASE) are low, which means a good quality of predictions. The nugget-to-sill ratio presented in Table 1 indicated weak spatial dependence for almost all soil properties, which reflects high variance at short distances (high heterogeneity). Moderate and high spatial dependence were obtained for clay and hydraulic conductivity, respectively. Cambardella et al. (1994) suggested that there is weak spatial dependency if the ratio nugget-to-sill was >0.75. According to the same author, weak spatial dependence is controlled by non-intrinsic changes in soil such as land-use management practices (e.g., fertilizers application and tillage).

In this sub-watershed, soils were mostly sandy loam soils that were formed mainly with sand (62.7 %), followed by silt (22.0 %) and clay (15.3 %), with a relativity low average of organic matter (3.6 %) and moderate to fast hydraulic conductivity (1.5–15.1 cm h^{-1}). Sandy soils are more prone to achieve the wilting point during drought periods due to their low water-holding capacity. The associated K factor values were predicted to vary from 0.0026 to 0.047 t ha h ha^{-1} MJ^{-1} mm^{-1}, with a mean value of 0.023 t ha h ha^{-1} MJ^{-1} mm^{-1}.

Prediction maps of soil properties and the respective soil erodibility are shown in Fig. 2. The maps show some trends. Soils with low sand contents were located mainly in the north and south parts. Clay percentages show a high spatial variation, with highest values in some zones in the north and center. Silt percentage has a strong negative correlation with sand percentage because the areas with higher sand were associated with areas with lower silt contents. The modified silt prediction map (silt and very fine sand, the most erodible particles) reveals identical trends to the silt prediction map. Thus, the most erodible particles were more likely to occur in the North and South/Southwest.

Soil hydraulic conductivity is a fundamental parameter to understand flow process in soils. The highest hydraulic conductivity, i.e., permeability, occurs in the central and east parts, close to the water (more than 4 mm min^{-1}), where soils generally have the highest values of sand and the lowest values of clay.

Through the prediction map of soil erodibility (K), it could be seen that the highest soil erodibility values were mainly located in the south and north sections, where the highest amounts of susceptible particles (silt and very fine sand) were found. Soils with high permeability are more resistant to erosion (less erodibility), and this was confirmed by the results.

Slope Length and Steepness Factor (LS)

The first step to determining the LS factor was the creation of the DEM (Fig. 3). The elevation in this area ranges between 136 and 215 m, and the highest values were in the northwest part, with values decreasing as the distance to the reservoir (situated at the east side) reduces. Slope values in this study area, also shown in Fig. 3, were found

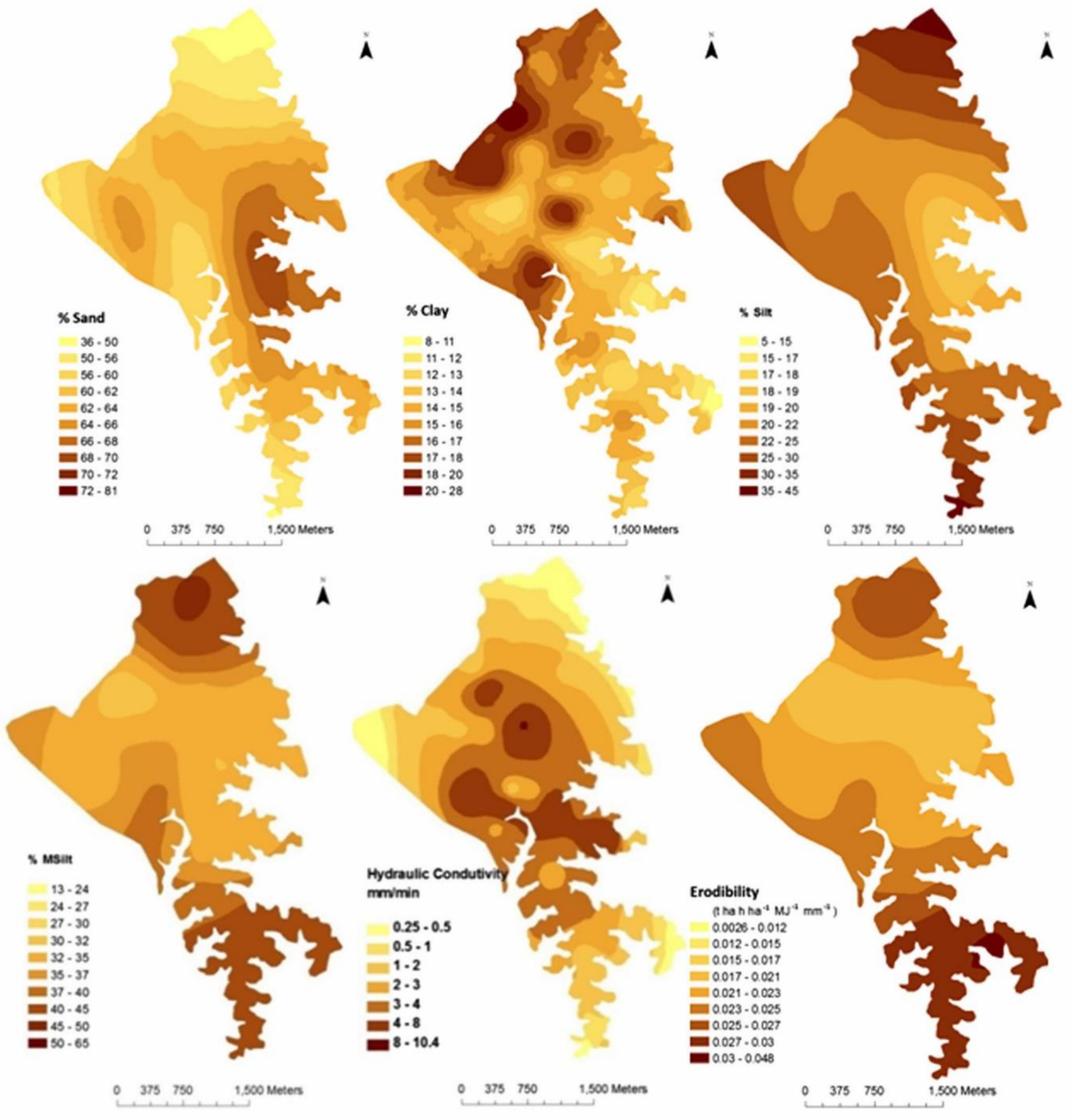

Fig. 2 Prediction maps of soil properties (texture, hydraulic conductivity) and soil erodibility (K factor)

to vary from 0 to 30 %, with an average of 7.5 %, whereas only 7.8 % of the total area exceeds a slope of 15 %. The highest slopes result in an increased overland flow, rilling, and concentrated flow depth. LS factor values (without units) in the study area (Fig. 3), which depend on slopes and flow accumulation, vary from 0 to 12.35, with a mean and standard deviation (SD) of 1.28 and 1.45, respectively. The outcomes reveal that the LS factor has a clear correlation with slope because areas with highest slope values have the highest LS factor values. The highest values of LS occur in the center of the area, in the south-west part, and the east side, close to the reservoir. These areas require special attention in the implementation of some land-uses, especially cultivation practices that make soil more susceptible to erosion (high erodibility) and with low vegetation cover. Increases in the LS factor can

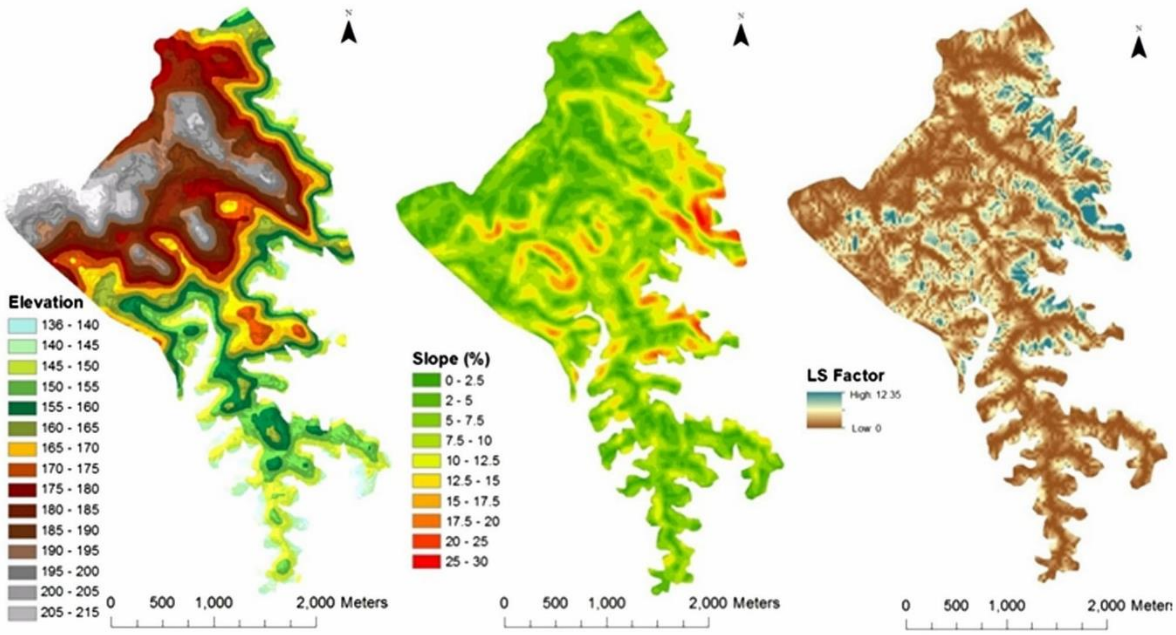

Fig. 3 Digital elevation model (DEM), the slope map, and the slope length (LS) map of the study area

produce higher overland flow velocities and correspondingly higher erosion (Van Remortel et al. 2004).

Rainfall Erosivity (R)

Rainfall erosivity values were estimated for 25 stations in the surrounding area of the Alqueva using daily precipitation values during 30 years (1979/80–2008/09); geostatistic maps (a continuous surface) were obtained using the OK interpolation method. These maps were used to estimate soil erosion in the study area.

Table 2 shows the estimated monthly and seasonal erosivity values in the Reguengos Meteorological Station (the nearest to the study experimental site) obtained through Eq. (4) and the mean values of erosivity for the Roncão study area achieved by geostatistics. According to the results, the rainfall erosivity in the study area was characterized by a strong seasonality effect, and the highest rainfall erosivity values were associated with the first rainfall events in autumn (433.6 MJ mm ha^{-1} h^{-1}), and the lowest values occur in the summer (80.4 MJ mm ha^{-1} h^{-1}). In the winter rainfall erosivity decreases slightly compared with autumn (196.6 MJ mm ha^{-1} h^{-1}), and in spring the decrease continues (143.4 MJ mm ha^{-1} h^{-1}). The annual erosivity, in the study area, was found to be 854 MJ mm ha^{-1} h^{-1}. In percentages, about 50.8 % of annual rainfall erosivity occurs in the autumn, 23.0 % in winter, 16.8 % in spring, and only 9.4 % in summer. These values and trends are comparable to the results obtained in other studies in Mediterranean regions (Van der Knijff et al. 1999; Diodato 2005) and the Iberian Peninsula (Panagopoulos and Antunes 2008; Angulo-Martínez and Beguería 2009; Beguería et al. 2009).

Vegetation Cover

Using satellite images, we obtained NDVI values. Figure 4 shows the spatial distribution, in the study area, of NDVI mean values for each season. NDVI in this agro-forestry watershed, during these 3 years (2006/07–2008/09), were found to have the highest values during the winter season with a mean of 0.457 and SD of 0.085. Availability of water in the soil and moderate temperatures in the winter create good conditions which facilitate natural vegetation growth. In the summer and in the autumn seasons, the NDVI values were the lowest with an average value of 0.099 (SD = 0.043) and 0.104 (SD = 0.057), offering less protection against storms. Spring shows NDVI between summer and winter values, with a mean of 0.219 (SD = 0.058). It shows that the vegetation in the wet period was denser than in the dry period, which is typical in Mediterranean regions (Van der Knijff et al. 1999). Therefore, the vegetation cover was clearly correlated to seasonal changes. Analyzing the maps, it can be seen that the highest values of NDVI were mainly located in the

Table 2 Monthly rainfall and monthly and seasonal erosivity (R) in the Reguengos meteorological station and for the study area

Season	Months	Reguengos meteorological station				Study area (Roncão)	
		Monthly rainfall (mm)		R factor (1979/80–2008/09)		R factor (geostatistic values) (1979/80–2008/09)	
		(1979/80–2008/09)	(2006/07–2008/09)	Monthly (MJ mm ha^{-1} h^{-1} $month^{-1}$)	Seasonal (MJ mm ha^{-1} h^{-1} (3 months)$^{-1}$)	Monthly (MJ mm ha^{-1} h^{-1} $month^{-1}$)	Seasonal (MJ mm ha^{-1} h^{-1} (3 months)$^{-1}$)
Autumn	October	73.0	72.8	149.6	**387.2**	154.4	**433.6**
	November	68.8	45.4	131.5		160.1	
	December	73.0	30.0	106.1		119.1	
Winter	January	58.1	58.4	95.8	**200.2**	93.3	**196.6**
	February	42.4	50.9	50.9		57.3	
	March	37.6	23.8	53.5		46.0	
Spring	April	52.4	50.4	62.4	**126.4**	70.6	**143.4**
	May	42.6	37.4	44.4		52.2	
	June	14.2	12.5	19.6		20.6	
Summer	July	4.6	0.7	3.4	**77.1**	3.4	**80.4**
	August	5.1	4.6	9.2		6.8	
	September	27.2	15.7	64.5		**70.2**	
Annual		498.9	401.9		**790.9**		**854.0**

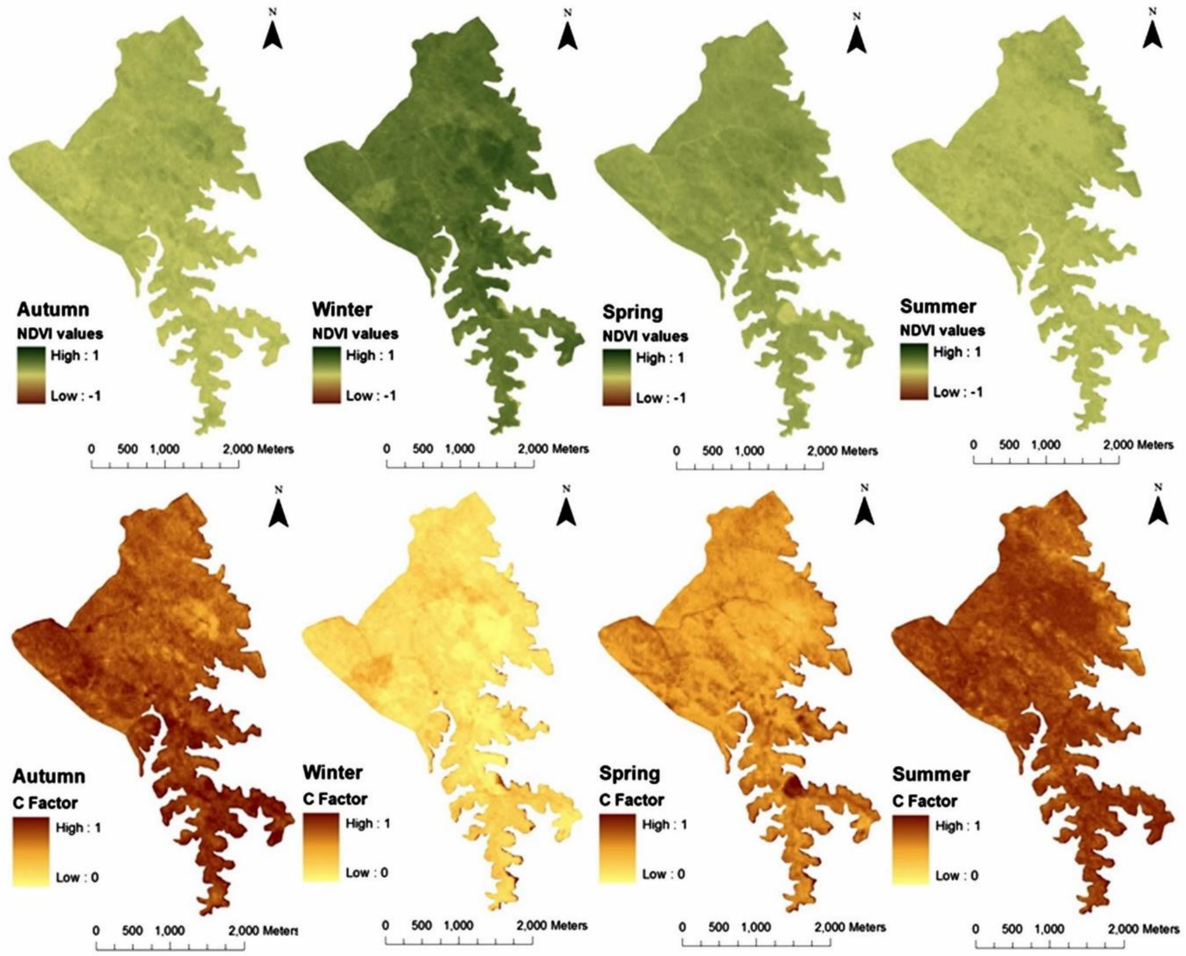

Fig. 4 Spatial distribution of Normalized Difference Vegetation Index (NDVI) values and corresponding C factor values of the study area

North section, where there were more forested areas as observed in the field.

Looking at C factor maps (Fig. 4), it can be concluded that C factor values have a negative correlation with NDVI values and that the highest values of NDVI caused the lowest values of the C factor, resulting in the lowest erosion values. On the other hand, the lowest NDVI values coincide with the highest C values, and that means more erosion. The results revealed that C factor values in winter (with a mean of 0.193 and SD of 0.114) were lower than in summer (with a mean of 0.795 and SD of 0.079) and autumn (with a mean of 0.789 and SD of 0.113). For spring, the values were found to be in the middle with a mean of 0.563 (SD = 0.099). These results can be explained with the type of vegetation in this experimental area, which consists of an agro-silvo-pastoral system in transition toward a silvo-pastoral, with evergreen trees and scrubs, and natural pasture that dried out or died in the summer and showed growth in the winter, due to climate conditions.

Seasonal Soil Loss

All factors were integrated within the raster calculator option of the ArcGIS Spatial Analyst tool to obtain and quantify soil erosion rates for each season. The spatial distribution of soil erosion estimated for each season is shown in Fig. 5. The maps show the average values per season in t ha^{-1}. Throughout the prediction maps several variations can be seen. The highest values of soil loss were more likely to occur in autumn (16.9 % of the area with soil loss higher than 20 t ha^{-1}), when the rainfall erosivity reaches the peak due to heavy rainstorms. During this period, the soil is very susceptible to erosion since vegetation cover (natural) was still low–moderate after the dry summer with high temperatures. On the other hand, winter

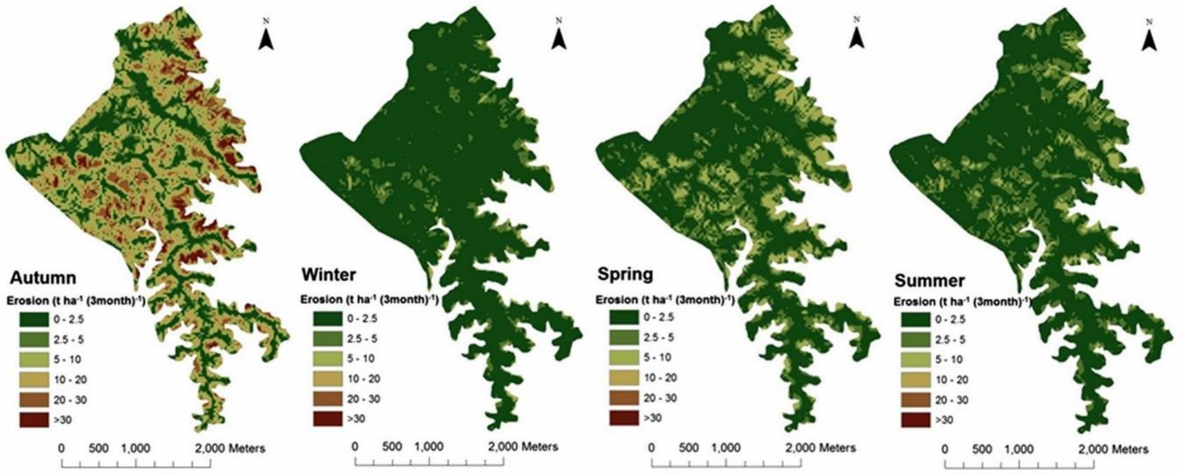

Fig. 5 Prediction maps of the seasonal soil erosion of the study area

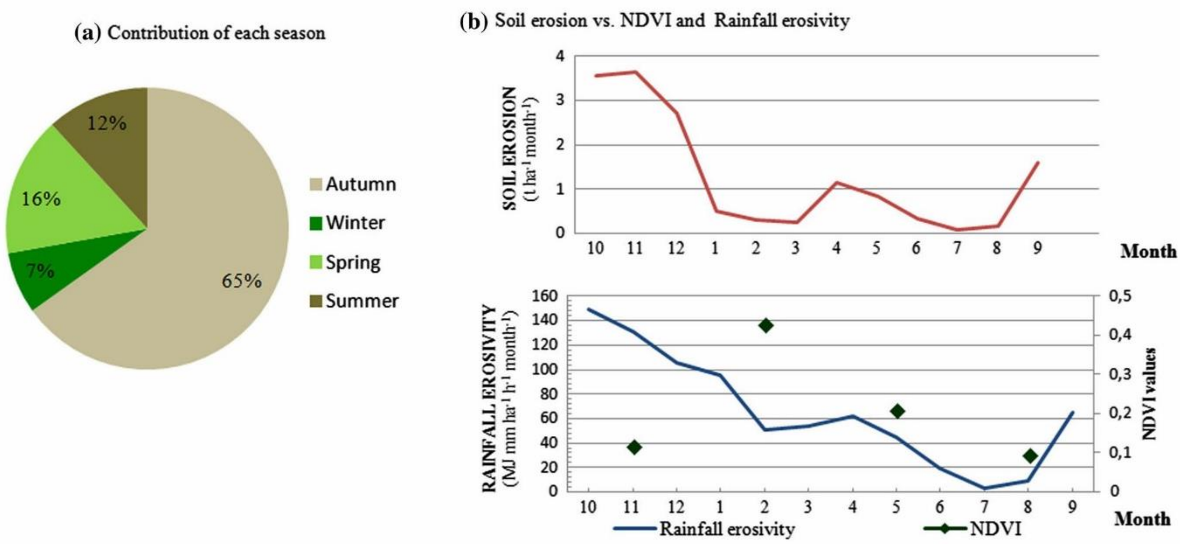

Fig. 6 Seasonality of soil erosion: **a** the contribution of each season to annual soil erosion and **b** annual distribution of soil erosion versus NDVI values and rainfall erosivity

shows the lowest erosion (only 0.23 % of the area with soil loss higher than 10 t ha^{-1}) especially because soil was more protected due to high vegetation cover. In the spring and summer, despite the fact that the rainfall erosivity values were the lowest, soil erosion was higher than in the winter, since the vegetation cover was inferior.

The contribution of each season to the total annual soil erosion is shown in Fig. 6a. Autumn was the season that contributes the most to annual erosion (with 65 %), followed by spring (16 %) and summer (12 %), with less impact in the winter (7 %). The maximum variation in soil erosion between seasons was 58 %. This can be explained by the difference between rainfall erosivity and vegetation growth (shown more clearly in Fig. 6b). In autumn, the surface was exposed to intensive rainfall episodes after a long dry period that leaves soil with low vegetation.

To better identify the critical periods and the importance of seasonal variations of some factors already referred, NDVI values and rainfall erosivity can be examined in the Fig. 6b is shown the temporal profile of soil erosion prediction. In the chart, the mean values of NDVI were presented per season (the mean value for the 3 years), due to difficulty in obtaining it per month, as already explained. Rainfall erosivity and predicted erosion were presented per

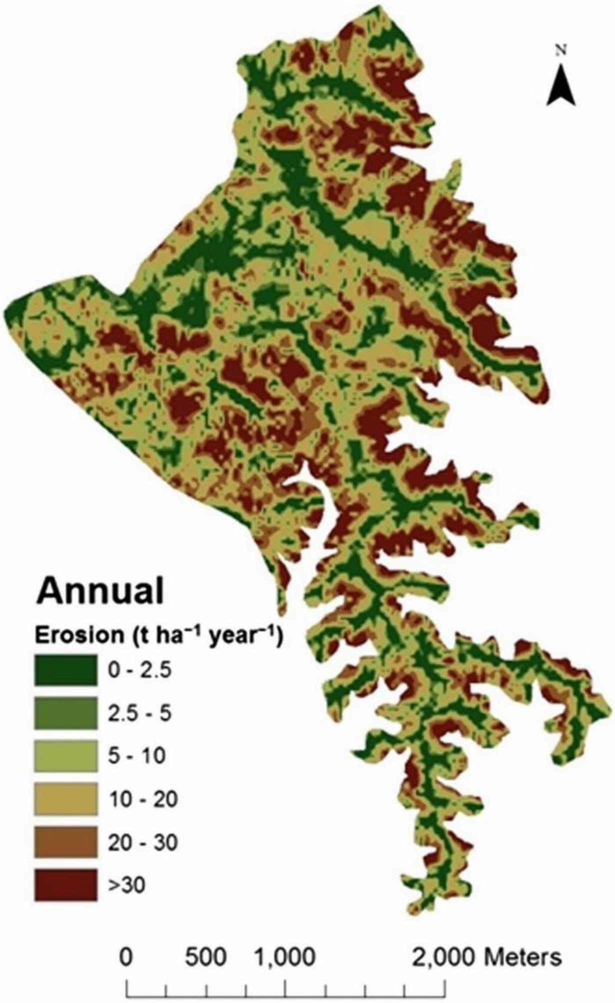

Fig. 7 Prediction map of the potential annual soil erosion

month. As can be observed, there was a great correlation especially between rainfall erosivity and soil loss. Vegetation (in that case analyzed through NDVI values) also has an influence, and it is noticeable in the first months of the year because the high vegetation cover contributes to decrease soil erosion, despite the fact that rainfall erosivity is still high. On the other hand, particularly in October and November, the highest rainfall erosivity and low vegetation cover reflect the highest erosion values, which exceeds 3.5 t ha^{-1} $month^{-1}$ in these months, in the experimental area.

Some studies have already evaluated these seasonal variations (Van Leeuwen and Sammons 2003). However, these are especially notable in the Mediterranean regions that have particular climatic conditions (Van der Knijff et al. 1999). Panagos et al. (2011) obtained similar trends for the Strymonas river basin (SE Europe), observing the highest soil erosion during October and November, especially for forest, scrublands, and natural grasslands. Climate change could possibly intensify the differences between seasons and increase the frequency of heavy rainstorms throughout the year.

Annual Soil Loss

An annual map of soil loss was generated through the sum of season values (Fig. 7). The mean of annual erosion in this area was found to be 15.1 t ha^{-1} $year^{-1}$ in the period studied (2006/07–2008/09). The terrain with serious erosion risk (greater than 30 t ha^{-1} $year^{-1}$) covers about 16.7 % of the area. On this map, it can be seen that these high soil erosion risk values lie mostly in the southwest part of the experimental area, and they were associated mainly with greater slopes (see Fig. 3) and areas with low vegetation cover (Fig. 4). In addition to those, soils in this zone have high soil erodibility factor values (Fig. 2). These results demonstrated that the soil erosion is highly dependent on the local terrain, soil properties, and land-use. Areas with high susceptibility of soil conditions (soil erodibility and topography) would need special priority for implementation of soil conservation practices.

The estimated values of soil erosion in the present study are comparable to the predicted and measured values for other Mediterranean watersheds. On the Greek part of the Strymonas river basin, the mean annual erosion was estimated to be 12.09 t ha^{-1} in forest and natural land-uses, using an approach based on the inheritance of USLE-family models and considering monthly variation (Panagos et al. 2011). In the southern Spain, the erosion rate measured on shrublands was between 0.5 and 21.5 t ha^{-1} - $year^{-1}$ with values of 8 t ha^{-1} $year^{-1}$ for average annual precipitation similar to Alqueva (Kosmas et al. 1997).

The results were also verified by erosion figures derived from previous studies in the area. These values are similar to the estimated values with the PESERA model (Kirkby et al. 2004), whose obtained values were between 0 and 32 t ha^{-1} $year^{-1}$, with high values in the center part of the experimental area. Compared to the values measured at the Vale Formoso Experimental Center (near the Alqueva reservoir area), using experimental plots, the annual average for wheat-culture, plowed field, stubble field, and pasture were 7.6, 3.1, 4.4, and 0.2 t ha^{-1}, respectively (Roxo and Cortesão Casimiro 1997). These measured values were lower than the estimated values in the study area (agro-silvo pastoral system), which can be explained by different methods, area size, topography, and soil characteristics.

The RUSLE equation estimates only local erosion amounts and cannot be used to estimate the sediment yield for an entire study area, which is possible using the sediment delivery ratio (SDR). SDRs are used to estimate a percentage of the total eroded sediment that is being effectively transported from its source into the fluvial system. The annual sediment and yield uses the SDR curve developed by Vanoni (1975):

$$\mathrm{SDR} = 0.4724\, A^{-0.125}, \tag{7}$$

where A is the area of the basin (km^2). Using the equation, for the experimental area of Roncão (782 ha), the SDR is 0.37, and the estimated amount of annual sediment is approximately $4{,}314 \times 10^3$ kg. Assuming similar soil erosion for all the surrounding Alqueva area (5.5 % of the total influence area that comprises territory in Portugal and Spain) ($A = 301{,}318$ ha, $\mathrm{SDR} = 0.17$), the amount of annual sediment is estimated to be around $789{,}648 \times 10^3$ kg for this watershed. However, the SDR, according Jain and Kothyari (2000), is a spatial phenomenon that varies with watershed heterogeneity (such as topography, land-use, soil characteristics, and rainfall erosivity) and the SDR-area relationship does not take into account it. So, these estimations are an indicator of sediment yield and this is a limitation of the study. For accurate results, it is important to discretize a catchment into sub-areas each having approximately homogeneous characteristics and uniform rainfall distribution (Jain and Kothyari 2000).

Sustainable Land Management (SLM)

The study results show the importance of focusing on seasonal climate conditions when selecting strategies to control soil erosion. The seasonal soil degradation maps can be used to determine the optimal balance between rainfall and vegetation cover, essential to reduce soil erosion during the year. One study documented in the WOCAT database (Schwilch et al. 2012) describes the implementation of a sustainable land measure on a silvo-pastoral system in Italy (Rendina Basin), accounting for the seasonality of the Mediterranean region. The farmers implemented a controlled grazing in deciduous woods that was seasonally limited in summer when grass cover in rangeland suffers water stress and the overgrazing can lead to land-degradation impacts. Additionally, many other examples of SLM technologies and approaches, which are being implemented in Mediterranean regions, are related to the WOCAT database. In the north of Portugal, a strip network system for fuel management was implemented facing specific land degradation due to forest fires (Schwilch et al. 2012).

Future work aims to study soil erosion of more land-uses that are being implemented in the Alqueva watershed due to increased water availability. Consequently, the prediction maps produced will be used as a solid base to use in collaboration with WOCAT methodology to analyze promising approaches and technologies, so as to provide SLM options for decision-makers. Furthermore, it can help through the estimation of the optimal alternatives for various land-uses according to the specific characteristics of the terrain and the definition of soil conservation practices in possible, different land-uses. Through the WOCAT decision-support tool, involving collaboration with both local people and governmental institutions, it will be possible to identify, assess, and select the best practices for this watershed, allowing the delineation of seasonal strategies for different land-use scenarios, in order to reduce soil erosion rates. Additionally, these seasonal soil erosion results in this Mediterranean area will contribute to the documentation of SLM technologies and approaches (case studies) that will be part of WOCAT databases and will contribute for understanding patterns for areas with similar climate conditions. These new approaches could decrease the risk of soil erosion, reducing costs, increasing the reservoir's life span, and promoting the sustainability of these ecosystems.

Conclusions

Updating the RUSLE model with the intra-annual variations of rainfall erosivity (R) and vegetation cover (C) factors, the soil erosion rates for each period were estimated and the seasonality existing in the Mediterranean regions was confirmed. Soil erosion is a seasonal phenomenon greatly affected by changes in rainfall and vegetation cover during the year. When peak rainfall erosivity coincides with low vegetation cover, soil erosion risk is increased considerably. The greatest values in this area are predicted to occur during the autumn season, and the lowest values are predicted to occur during the winter. The present study provided useful information not only about critical seasons but also about current local conditions. The results showed serious erosion risk for areas with the highest slopes, and the maps allow the identification of these "hotspots."

Planning of soil conservation measures, especially concerning agricultural practices, requires a good knowledge of all factors affecting soil erosion and their spatial and temporal variability. This study demonstrates the significance of incorporating temporal variability of some factors when modeling soil erosion. Therefore, understanding seasonal variations would be essential to delineate appropriate strategies of SLM for the watershed studied and cope with climate changes.

As well as safeguarding the future of the reservoir, this work also aims to contribute more broadly to the conservation of soil and water resources in the Alqueva region. This RUSLE seasonal-updating model will be used to obtain prediction maps for different land-uses, rendering it a good tool to integrate with WOCAT methods to support the decision-making on SLM in the Alqueva watershed area.

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Appendix IV

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Spatial variability of soil properties and soil erodibility in the Alqueva reservoir watershed

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Abstract. The aim of this work is to investigate how the spatial variability of soil properties and soil erodibility (K factor) were affected by the changes in land use allowed by irrigation with water from a reservoir in a semiarid area. To this end, three areas representative of different land uses (agroforestry grassland, lucerne crop and olive orchard) were studied within a 900 ha farm. The interrelationships between variables were analyzed by multivariate techniques and extrapolated using geostatistics. The results confirmed differences between land uses for all properties analyzed, which was explained mainly by the existence of diverse management practices (tillage, fertilization and irrigation), vegetation cover and local soil characteristics. Soil organic matter, clay and nitrogen content decreased significantly, while the K factor increased with intensive cultivation. The HJ-Biplot methodology was used to represent the variation of soil erodibility properties grouped in land uses. Native grassland was the least correlated with the other land uses. The K factor demonstrated high correlation mainly with very fine sand and silt. The maps produced with geostatistics were crucial to understand the current spatial variability in the Alqueva region. Facing the intensification of land-use conversion, a sustainable management is needed to introduce protective measures to control soil erosion.

1 Introduction

Soil erosion is a significant economic and environmental problem worldwide as a driving force affecting landscapes (Zhao et al., 2013). It is a very dynamic and complex process, characterized by the decline of soil quality and productivity, as it causes the loss of topsoil and increases runoff (Lal, 2001; Yang et al., 2003). Furthermore, soil erosion often causes negative downstream impacts, such as sedimentation in rivers and reservoirs, decreasing their storage volume as well as lifespan (Pandey et al., 2007; Haregeweyn et al., 2013).

One of the main causes of soil loss intensification around the world is associated with land-use change (Leh et al., 2013). The relationship between different land use and soil susceptibility to erosion has attracted the interest of a variety of researchers (Yang et al., 2003; Cerdà and Doerr, 2007; Blavet et al., 2009; Biro et al., 2013; Wang and Shao, 2013), who have shown the impact of changes in vegetation cover and agricultural practices on soil properties and therefore on overland flow. Generally, cultivated lands experience the highest erosion yield (Cerdà et al., 2009; Mandal and Sharda, 2013). In the Mediterranean regions, in combination with these anthropogenic factors, climate change has amplified the concerns about soil erosion since it is expected that there will be an increase of dry periods followed by heavy storms with concentrated rainfall (Nunes et al., 2009).

Some models have been developed to predict soil loss and sediment delivery. The Revised Universal Soil Loss Equation (RUSLE) is the most used empirical equation for modeling annual soil loss from agricultural watersheds (Renard et al., 1997). The susceptibility of soil erosion and land degradation depends largely on various inherent soil properties, namely chemical, physical, biological and mineralogical properties (Cambardella et al., 1994; Pérez-Rodríguez et al., 2007). However, according to the RUSLE model only some

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of the soil's properties define soil erodibility (K factor), such as particle-size composition, the content of organic matter, soil structure and permeability. Therefore, the K factor is the most used and is an important index to measure soil susceptibility to erosion (Panagopoulos and Antunes, 2008).

Spatial variability in soils occurs naturally as a result of complex interactions between geology, topography and climate. Moreover the spatial variability of soil properties, which influence soil susceptibility to erosion, is highly related to anthropogenic factors particularly in cultivated lands (Paz-González et al., 2000; Wang and Shao, 2013). Thus, information on the spatial variability and the interactions between soil properties is essential for understanding the ecosystem processes and planning sustainable soil management alternatives for specific land uses (Pérez-Rodríguez et al. 2007; Ziadat and Tamimeh, 2013).

Classical statistics and geostatistics methods have been widely applied in studies about spatial distribution of soil properties (Pérez-Rodríguez et al., 2007, Tesfahunegn et al., 2011). Geostatistical techniques based on predictions and simulations have been used to describe areas where predicted information is established by a limited number of samples (Goovaerts, 1997). Geostatistics provides tools for analyzing spatial variability structure and distribution of soil properties and evaluating their dependence (Panagopoulos et al., 2014).

The biplot methodology provides an added value for analyzing spatial variability of soil properties. This multivariate statistical technique allows the graphical representation of a large data matrix (Gabriel, 1971), whereby it is possible to interpret the relations between individuals (samples) and between variables, as well as between both. Biplot can also indicate clustering of units with close characteristics, showing inter-unit distances as well as displaying variances and correlations of the variables (Gallego-Álvarez et al., 2013). The HJ-Biplot permits not only the analysis of the behavior by sample but also the determination of which variable is responsible for such behavior (Garcia-Talegon et al., 1999), allowing a visual appraisal to establish relations between soil properties and land uses.

The construction of the Alqueva dam in a semiarid area of southern Portugal created one of the largest artificial lakes in Europe. Taking advantage of water availability from the reservoir, this Mediterranean region has been subjected to land-use conversion from the native montado grassland to intensive agricultural uses. Land-use conversion from the native ecosystem to agriculture may alter physical, chemical and biological soil properties, which consequently may increase soil erosion and siltation in the reservoir. Soil erosion in the area has to be carefully evaluated in order to undertake sustainable soil management measures. Therefore, the aim of this study was to evaluate the effects of cultivation practices on some chemical and physical soil properties and on soil erodibility (K factor on RUSLE), and to characterize their spatial variability using geostatistics and HJ-Biplot methodology.

2 Material and methods

2.1 Study area

Located in the semiarid Alentejo region of Portugal, at the Guadiana River, the Alqueva reservoir (8°30′ W, 38°30′ N) covers an area of 250 km^2, and the capacity of the reservoir is 4.15 km^3. The main arguments for the implementation of what is considered the largest artificial lake in Europe were based on the need to combat the growing effects of desertification and to prevent the annual and monthly fluctuations in precipitation. One of the main goals of the Alqueva Multipurpose Project was the implementation of 120 000 ha of newly irrigated land in the Alentejo. The Alentejo region, covering an area of 27 000 km^2, is considered one of the most depressed regions of the European Union and characterized by a Mediterranean climate with very hot and dry summers and mild winters. The average temperature ranges from 24 to 28 °C in hot months (July/August) and from 8 to 11 °C in cold months (December/January). The average annual precipitation at the nearest meteorological station, for the last 30 years, is 517.2 mm. The region is affected by intense dry periods followed by heavy, erosive rains concentrated in the autumn season.

The study experimental site (farm "Herdade dos Gregos"), located in the surrounding area of the reservoir (Fig. 1), is a private property with 900 ha. The landscape is characterized by its hilly topography with significant altitude variations (mainly between 100 and 250 m). The bedrock of the study area is rocky, and, according to the World Reference Base for Soil Resources (FAO, 2006), the two types of soil in this area are Haplic Luvisols (LVha) and Lithic Leptosols (LPli). This farm was selected to include a diversity of land uses, including native montado grassland and more intensive land uses, with irrigation, namely olive tree orchard and lucerne cultivation. Direct pumping from Alqueva reservoir is done on this private property since it is near the reservoir.

The typical landscape in the Alentejo region is the montado native grassland, an agrosilvopastoral system characterized by savannah-like, low-density woodlands with evergreen holm oaks (*Quercus ilex*). For that reason, an area of the montado grassland (20.7 ha), used as a permanent pasture for the cattle, was selected for this study. This small area is located in the high altitudes of the "Herdade dos Gregos" (from 200 to 240 m) with a slope that varies from 1.4 to 20.9 %. Tillage (at about 15 cm depths) was done only once every 10 years to decrease shrub competition (the most recent one was 4 years before the study implementation), and the soil is not subjected to any fertilizer. Four years before the study implementation, there was a fire on this agrosilvopastoral area of the farm.

Taking advantage of the water availability, another land use (with 33.5 ha) is an irrigation area (pivot sprinkler irrigation system) on which lucerne (*Medicago sativa*) is sown four times a year. Lucerne, once dried, is nutritional for cat-

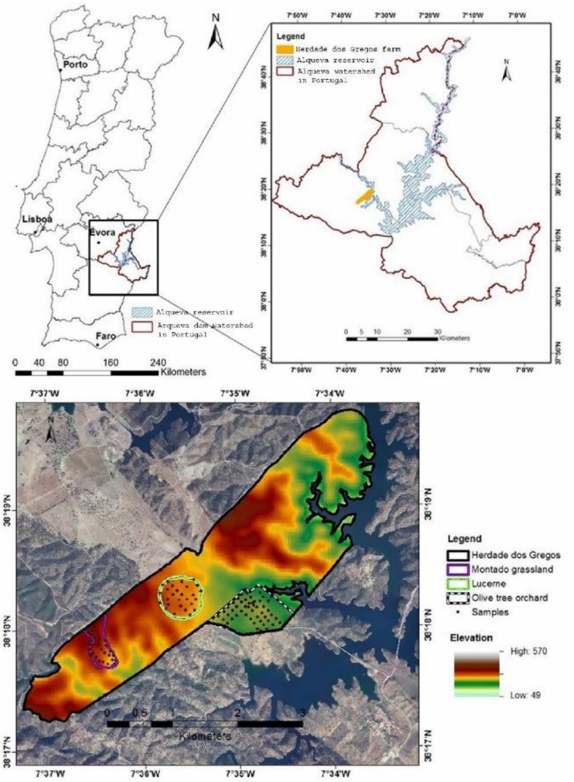

Figure 1. Location of the study area at the Alqueva dam watershed in Portugal.

tle, and it incorporates nitrogen in the soil. In this area, conventional tillage is used, involving multiple aspects: plough (about 20 cm depth) in fall, fallowing cultivator (about 15 cm depths) and disc harrow (about 10 cm depths) subsequent to soil tillage. Inorganic fertilizers were applied to the cultivated field at a rate of 100 kg NPK ha^{-1}. This land use is placed in the midland (194–220 m), and the slope varies from 0 to 9 %.

Another irrigated land use consists of an olive tree plantation (57.5 ha), which is done in strips. This cultivation has a drip irrigation system, is fertilized once every 2 years and is ploughed once a year to decrease weed competition. The olive orchard is located in the low elevations of the farm (150–186 m), and it is on the side of the reservoir (Fig. 1). The slope varies from 0 to 14.2 %.

2.2 Soil sampling and laboratory analysis

Since the objective was to study the relation between soil properties and the K factor from RUSLE, the soil samples were collected from 0 to 20 cm depth, according to Renard et al. (1997). In order to predict variations in short distances, 25, 27 and 52 soil samples were randomly collected respectively in montado, lucerne and the olive orchard (see Fig. 1). Samples were air-dried and then dried for about 6 h at 40 °C on a ventilated oven, and they were passed through a 2 mm sieve to remove rocks and gravel. The particle-size distribution was determined by the Bouyoucos hydrometer method (Bouyoucos, 1936). Soil organic matter content was determined using the Walkley and Black (1934) method, a wet oxidation procedure. The soil's total nitrogen content was determined according to Kjeldhal digestion, distillation and the titration method (Bremmer and Mulvaney, 1982). Soil pH and electrical conductivity were measured with a glass electrode in a 1 : 2.5 soil–water suspension (Watson and Brown, 2011).

2.3 Soil erodibility factor

Soil erodibility factor (K) (Mg ha h ha^{-1} MJ^{-1} mm^{-1}) was estimated using soil property values – such as particle-size composition, content of organic matter, soil structure and permeability – in the 104 sample points described above. This factor represents the soil-loss rate per erosion index unit for a specified soil as measured on a standard plot (Renard et al., 1997). An algebraic approximation of the nomograph (Eq. 1) was used to estimate the K factor (Renard et al., 1997):

$$K = [2.1 \times 10^{-4}(12 - \mathrm{OM}) \times M^{1.14} + 3.25(s-2) + 2.5(p-3)]/759, \quad (1)$$

where OM is the percentage of organic matter, s is soil structure class, p is permeability class and M is the product of the percentage of modified silt (silt particles and very fine sand) or the 0.002–0.1 mm size fraction and the sum of the percentage of silt and percentage of sand. K is expressed in SI units of Mg ha h ha^{-1} MJ^{-1} mm^{-1}. To estimate the permeability, the field-saturated hydraulic conductivity was measured in the field using a double-ring infiltrometer (six site measurements per land use, each one with five repetitions). Permeability class and soil structure class were defined in accordance with Renard et al. (1997).

2.4 Statistical and geostatistical analysis

Data were subjected to classical analysis using SPSS 17.0 software to obtain descriptive statistics, namely the mean, minimum and maximum; standard deviation (SD); coefficient of variation (CV); and skewness of each parameter.

Soil data were introduced in the ArcGIS environment, and geostatistical analyses were performed using Geostatistical Analyst tool, in other to examine spatial distribution of soil properties. Prior to using geostatistics to obtain prediction maps, a preliminary analysis of data was done to check data normality and global directional trends. Skewness is the most common statistical parameter to identify a normal distribution that is confirmed with skewness values varying form −1 to +1. Data transformation to normal distribution was necessary for some soil properties, and geostatistical analysis tools were used (log or Box–Cox method). Trend analysis was performed to examine the presence of any global directional trend in our data, an overriding process that affects all measurements in a deterministic way (nonrandom). So, when necessary, the trend removal was done using geostatistical analysis tools to more accurately model the variation (Panagopoulos et al., 2006).

The geostatistical methodology is based on the creation of a semivariogram (SV), a graphical representation (Eq. 2) that describes how samples are related to each other in space, and it is based on

$$\gamma(h) = 1/2N(h) \times \sum [Z_i - Z_{(i+h)}]^2, \tag{2}$$

where $\gamma(h)$ is the variance (the most related samples have lower values of variance), $N(h)$ is the number of samples that can be grouped using vector h, Z_i represents the value of the sample and $Z_i + h$ is the value of another sample located at a distance $\|h\|$ from the initial sample Z_i (Chiles and Delfiner, 1999).

Ordinary Kriging (OK) was selected as a geostatistical method. OK is considered one of the most accurate interpolation techniques which assumes that variables close in space tend to be more similar than those further away (Goovaerts, 1999).

Using the Geostatistical Analyst tool (ArcGIS) and selecting the OK methods, a semivariogram was created for each measured property. In the Kriging method different semivariogram models can be used (e.g., spherical, exponential) and the selection is usually performed by employing the cross-validation technique, which permits the evaluation of the prediction accuracy. Cross validation was executed to investigate the prediction performances through the statistical values, as the mean error (ME) or root-mean-square standardized error (RMSSE), which results from comparing the estimated semivariogram values and real observed values. Additional semivariogram parameters were analyzed to better understand the spatial structure and dependence of each variable. Nugget is the variance at distance zero and reflects the sampling error. Sill is the semivariance value at which the semivariogram reaches the upper bound and flattens out after its initial increase; it is the variance in which the samples are no longer spatially related to the study area.

Once the cross-validation process was completed, interpolation maps of spatial distribution, for each soil variable, were produced according to the semivariogram model selected, in the ArcGIS software.

2.5 HJ-Biplot

HJ-Biplot represents a matrix, without assumptions related to its probabilistic distribution, permitting a graphic representation of the geometric data structure, representing the data set (samples and variables) variability. The prefix "bi" is due to a simultaneous representation of the matrix rows and columns, searching for the maximum representation quality possible, at the same scale (Martín-Rodríguez et al., 2002; González-Cabrera et al., 2006; Gallego-Álvarez et al., 2013).

A data matrix $\mathbf{X}$ suffers a factorization to reduce its dimensionality through single-value decomposition, the algebraic base of biplot representation (Gabriel, 1971):

$$\mathbf{X}_{(n\times p)} = \mathbf{U}_{(n\times r)} \mathbf{\Lambda}_{(r\times r)} \mathbf{V}'_{(r\times p)}, \tag{3}$$

where $\mathbf{\Lambda}_{(r\times r)}$ is a diagonal $(\lambda_1, \lambda_2, \ldots, \lambda_r)$ corresponding to the r eigenvalues of XX' or $\mathbf{XX}'$, $U_{(n\times r)}$ is an orthogonal matrix whose columns are the eigenvectors of $\mathbf{XX}'$ or $\mathbf{X}'\mathbf{X}$ and $U_{(n\times r)}$ is an orthogonal matrix whose columns are the eigenvectors of $\mathbf{XX}'$.

With the *MultBiplot software*, developed by the University of Salamanca (Vicente Villardón, 2014), an HJ-Biplot was used to determine the relation between soil properties and land uses, and the correlations between both (soil properties and land uses), thereby defining patterns and clustering the samples in groups.

On the HJ-Biplot graphic representation, the points represent individuals (samples) and the vectors represent variables (in this case, chemical and physical soil properties). To interpret and discuss the graphs obtained with this methodology, it is essential to be aware of the following (Gallego-Álvarez et al., 2013):

- The distance between points represents the variability and can be interpreted as similarity or dissimilarity; i.e., the close samples have similar behaviors.
- The angle formed by variable vectors is interpreted as correlation; i.e., small angles between variables represent similar behaviors with high positive correlations, and the obtuse angles that are almost a straight angle are associated with variables with high negative correlations; i.e., the cosine value of the angles represents the correlation between variables.
- The proximity of individual points and variable vectors means high preponderance; in other words the closer a point is to a variable vector, the more important this sample is to explain this variable.
- The length of the vector represents the variable's variability; the longer the vector, the higher this variability.

3 Results and discussion

3.1 Descriptive statistics

The descriptive statistics of soil properties are given in the first part of Table 1. All measured parameters varied considerably within the areas (different land uses) as indicated by the coefficient of variation (varies from 4.2 to 70.2 %). Nitrogen (N) and organic matter (OM) show the highest variation values, especially for cultivated fields (lucerne cultivation and olive orchard), which can be explained with the lack

Table 1. Descriptive statistics of soil properties and parameters of the fitted variogram models and the cross-validation results.

	Classic statistics					Geostatistics					
	Mean	CV (%)	Min	Max	Skewness	Variogram	Nugget	Sill	Nugget /sill	ME	RMSSE
	Montado grassland ($n = 25$)					Montado grassland ($n = 25$)					
Clay (%)	17.29	37.7	5.68	29.62	0.07	Exponential	0	38.30	0.00	0.0055	1.01
Silt (%)	29.55	17.2	12.99	39.72	−0.99	Exponential	0	36.00	0.00	0.0238	1.04
Sand (%)	53.16	13.5	39.68	70.34	0.33	Pentaspherical	0	57.60	0.00	0.0223	0.99
VFS (%)	11.13	25.6	4.49	19.04	0.16	Stable	0	12.00	0.00	−0.0188	0.99
OM (%)	5.22	32.1	2.25	10.35	1.19	Exponential*	0.031	0.07	0.44	−0.0003	1.04
N (%)	0.19	43.2	0.07	0.42	1.13	Exponential*	0.056	0.17	0.32	0.0001	1.04
EC ($dS\,m^{-1}$)	0.100	38.1	55.5	217.5	1.28	Exponential*	0.012	0.13	0.09	0.5640	0.95
pH	5.90	4.2	5.38	6.30	0.01	Exponential	0	0.06	0.00	0.0022	0.99
HC_{sat} ($cm\,h^{-1}$)	4.56	42.9	1.20	7.20	−0.57	–	–	–	–	–	–
K ($Mg\,ha\,h\,ha^{-1}\,MJ^{-1}\,mm^{-1}$)	0.021	31.4	0.006	0.039	0.43	Stable	0	0.001	0.00	0.0001	1.00
	Lucerne cultivation ($n = 27$)					Lucerne cultivation ($n = 27$)					
Clay (%)	13.29	28.8	5.65	22.28	0.32	Stable	0	15.30	0.00	0.0017	1.02
Silt (%)	33.79	26.6	8.35	47.29	−1.48	Stable*	0	44.20	0.00	0.0073	0.97
Sand (%)	52.93	17.7	39.32	79.99	1.00	Exponential	0	92.00	0.00	0.0297	0.98
VFS (%)	15.28	37.0	2.59	25.17	−0.39	Exponential	15.60	25.0	0.62	0.0347	1.04
OM (%)	2.08	52.8	0.45	5.44	1.21	Exponential*	15.90	119	0.13	0.0036	0.94
N (%)	0.11	70.2	0.02	0.35	1.43	Circular*	0.10	0.52	0.20	0.0017	1.01
EC ($dS\,m^{-1}$)	0.107	45.9	40.5	205.0	0.64	Exponential	1.15	1.79	0.64	0.2240	0.96
pH	7.14	4.3	6.53	7.85	0.02	Exponential	0.04	0.07	0.57	0.0052	1.07
HC_{sat} ($cm\,h^{-1}$)	5.95	26.7	0.65	1.30	−0.29	–	–	–	–	–	–
K ($Mg\,ha\,h\,ha^{-1}\,MJ^{-1}\,mm^{-1}$)	0.039	21.9	0.013	0.052	−0.88	Stable	0	0.01	0.00	0.0001	1.03
	Olive tree orchard ($n = 52$)					Olive tree orchard ($n = 52$)					
Clay (%)	9.83	28.8	5.40	16.66	0.52	Stable	0	8.04	0.00	0.0001	0.99
Silt (%)	24.37	46.8	3.82	43.36	−0.41	Pentaspherical	50.00	89.80	0.55	0.0001	0.90
Sand (%)	65.81	18.2	40.6	89.66	0.21	Exponential	0	16.10	0.00	0.0002	0.91
VFS (%)	18.14	32.5	4.49	19.04	0.16	Exponential	0.01	33.70	0.00	0.0037	1.05
OM (%)	2.10	52.8	0.62	8.35	3.54	Exponential*	0.07	0.16	0.44	−0.0006	1.02
N (%)	0.10	45.3	0.04	0.29	2.02	Exponential*	0.02	0.15	0.12	0.0028	1.10
EC ($dS\,m^{-1}$)	0.182	61.3	53.50	583.50	1.80	Exponential	0	1.4	0.00	0.6820	1.02
pH	5.48	7.6	4.30	6.21	−0.43	Exponential	0	0.21	0.00	−0.0002	0.95
HC_{sat} ($cm\,h^{-1}$)	2.60	64.9	0.00	0.67	−0.45	–	–	–	–	–	–
K ($Mg\,ha\,h\,ha^{-1}\,MJ^{-1}\,mm^{-1}$)	0.038	33.6	0.012	0.061	−0.36	Exponential	0.00	0.001	0.51	−0.0001	0.92

*Transformation for normal distribution.
CV – coefficient variation; Min – minimum; Max – maximum; VFS – very fine sand; N – nitrogen; OM – organic matter; EC – electrical conductivity; HC_{sat} – saturated hydraulic conductivity; K – soil erodibility; ME – mean error; RMSSE – root-mean-square standardized error.

of homogeneous fertilization or tillage practices applied to soil in these areas.

The skewness results, which vary from −1.48 to 3.54 in this study, indicated that some soil properties of the different uses were not normally distributed, especially OM and N. The principal reason for some soil properties having non-normally distributions may be related to soil management practices (Tesfahunegn et al., 2011). As already mentioned, data were transformed to normal distribution when necessary (see Table 1).

These mean results show significant differences between land uses for all the properties analyzed. From the particle size distribution reported in Table 1, the soils are mostly sandy loam, formed mainly of sand, followed by silt and low quantities of clay. However, there are some differences between land use areas that can be explained by soil type. The LPli soils are characterized by a thin layer (about 10 cm), in that case upon a schist rock, justifying the higher clay content at the montado grassland. The LVha soils in the lucerne cultivation and the olive orchard are characterized by a loam or sandy loam layer (first 20 cm) with good drainage over clay-enriched subsoil (upon a basic crystalline rock), explaining the lower values of clay and fine sand, especially in the olive orchard. Despite the same soil type, soil texture is different between lucerne and the olive orchard, which can be justified by land use. The lucerne is a more intensive cultivation (intensive irrigation, tillage and continuous cultivation; fertilizers and lime application), involving conditions that promote changes in the soil weathering and moisture and, consequently, in soil texture (Yimer et al., 2008). On the other hand the soil between olive trees is kept without vegetation for most of the year and can explain the clay drainage to a sub-layer.

Montado shows the highest content of OM (5.22 %), whereas lucerne and olive fields show the lowest values (2.08 and 2.10 %, respectively). Other studies suggest that OM is higher in no-tillage soils compared to minimum tillage that increases aeration (Celik, 2005). Tillage mixes the subsoil with topsoil; after soil erosion, the nutrients are easily leached and the surface becomes poor in nutrients (Al-Kaisi and Licht, 2005). As for OM, the highest value of N nutrient occurs in the montado (0.19 %) and the lowest values in lucerne (0.11 %) and the olive orchard (0.10 %), which is related to the tillage practice that is frequently employed in these last two land uses, while in the montado grassland the cattle enriches the soil.

Soil EC values (Table 1) were similar when comparing the montado grassland (0.100 $dS\,cm^{-1}$) and the lucerne field (0.107 $dS\,cm^{-1}$); they were slightly higher in the olive orchard (0.182 $dS\,cm^{-1}$) but not enough to raise salinity problems. Usually, the addition of fertilizers (that happens on lucerne and the olive orchard) can cause high EC due to the percentage of the salts which are leached by water irrigation (higher in the lucerne field).

The soil pH was significantly higher in the lucerne cultivation land (7.1) compared to the montado grassland (5.9) or in the olive tree orchard (5.5) (Table 1). The soil pH in the lucerne was greater due to lime application to increment the soil pH in that area. Lucerne's optimum pH for production is between 6.5 and 7.2, and lime application has been found to produce a significant improvement in nodulation of lucerne (both number and dry weight of nodules per plant) (Grewal and Williams, 2001).

Saturated hydraulic conductivity (HC) values were greater in the lucerne area (5.95 $cm\,h^{-1}$), slightly lower in the montado grassland (4.56 $cm\,h^{-1}$) and lowest in the olive orchard (2.60 $cm\,h^{-1}$). The lower permeability in the olive orchard can be explained by the clay-enriched subsoil or soil crust problems, and it may explain the higher values of EC, i.e., the greater concentration of salts. Also it can be explained by the frequency of tillage in the different land uses because aggregate stability and water infiltration rate are higher in soils subjected to limited tillage systems (Alvarez and Steinbach, 2009).

As a result, the K factor was different for the typical land use, montado grassland, compared to the lucerne cultivation and the olive orchard. The values increased with the intensification of the cultivation field, with the lowest values for montado grassland (0.021 $Mg\,ha\,h\,ha^{-1}\,MJ^{-1}\,mm^{-1}$) and the highest for the lucerne cultivation (0.039 $Mg\,ha\,h\,ha^{-1}\,MJ^{-1}\,mm^{-1}$) and the olive orchard (0.038 $Mg\,ha\,h\,ha^{-1}\,MJ^{-1}\,mm^{-1}$). Other studies had similar results, showing that the removal of permanent vegetation, the loss of OM and the reduction of aggregation, caused by intensive cultivation, contribute to decrease the K factor (Celik, 2005).

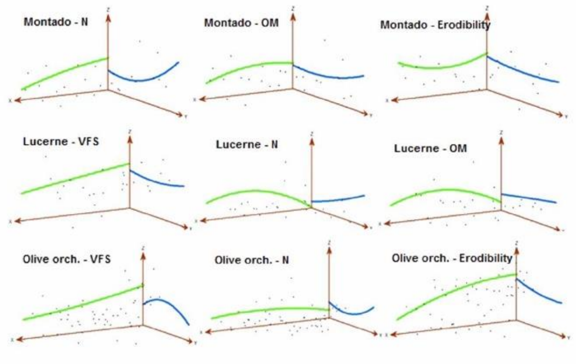

Figure 2. Three-dimensional perspective of the trends in the input data sets.

3.2 Spatial dependence of soil properties

Model selection for each soil property was based on the nugget, sill, ME and the RMSSE presented in the second part of Table 1 (Geostatistics).

Nugget is low in most soil properties studied, implying strong spatial dependence. The nugget-to-sill ratio is used to define spatial dependence of soil properties: if the ratio is < 0.25, there is strong spatial dependence; if it is 0.25 to 0.75, there is moderate spatial dependence; and if the ratio is > 0.75, spatial dependence is weak (Cambardella et al., 1994). As shown in Table 1 the ratio values indicate the presence of high to moderate spatial dependence for all soil parameters (values between 0 and 0.64). In general, there is stronger spatial dependence in montado (low nugget-to-sill ratio), which can be explained with the non-existence of extrinsic factors, such as management cultivation practices, that influence soil properties, and soil is left as it is for permanent pasture.

Cross validation facilitated the selection of the best-fit semivariogram for an interpolation map, which could provide the most accurate predictions. Closer values of the ME to 0, and closer values of the RMSS to 1, suggested that the prediction values were close to measured values (Wackkernagel, 1995). Most of the soil properties were best fitted with an exponential model, particularly in the montado area and olive orchard, whereas in lucerne the Gaussian, circular and stable semivariogram models were used.

3.3 Spatial distribution

The interpolation maps obtained with geostatistics are useful to better understand spatial variability and its influences. The variability of spatial soil properties can be influenced by natural factors (such as particle-size composition and topography) and anthropogenic factors (such as land cover or management practices) (Tesfahunegn et al., 2011). Sometimes, the effect of some factors is at least 1 order of magnitude

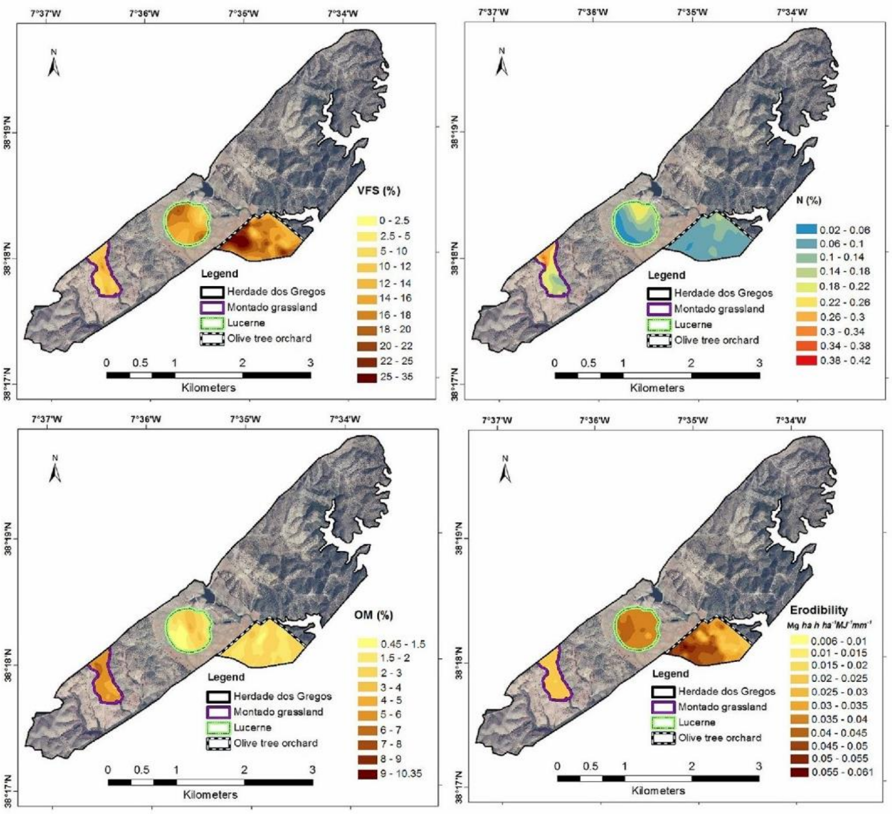

Figure 3. Prediction map of very fine sand (VFS), total nitrogen (N), organic matter (OM) and soil erodibility (K factor).

greater (as topography or soil type) than the land use. So, as mentioned trend analysis was performed to study the existence of directional trends caused by these factors with large scale of variation, and it is shown in Fig. 2. Global trend exists if a curve that is not flat (i.e., a polynomial equation) can be fitted to the data (for example for total N in montado or very fine sand (VFS) in the olive orchard). These trends were identified for part of the soil properties and for different land uses (Fig. 2). The strongest influence of a directional trend was identified from southeast to the northwest, which could be associated with the topography (Fig. 1) since the altitudes increase in accordance with these direction. So, trend removal is crucial to create more accurate prediction maps in order to justify an assumption of normality.

The interpolation maps for some studied soil properties are shown in Fig. 3. Through looking at the VFS distribution, it was noticed that the higher fractions of these particles (Fig. 3) were measured at low altitudes or on flat slopes such as the valley (see elevation in Fig. 1). This can be explained by erosion–deposition processes because these particles are easily detached and transported by water.

The highest percentages of N and OM were found on montado, as discussed previously. These two properties present similar distributions for all land uses. The nitrogen existing in the soil is mostly organic, and the inorganic forms (ammonium and nitrate) are easily leached or assimilated by plants. So, when OM breaks down due to mineralization, the N fraction decreases (Varennes, 2003). There were higher values in montado because the soil is not frequently tilled as it is in the other land uses. In the lucerne cultivation and the olive orchard, the variation of OM and N can be explained by inadequate management practices (e.g., inadequate fertilization rates, tillage, irrigation rates, seed rates).

Figure 3 illustrates the interpolation map for the K factor which was estimated through the Wischmeier nomograph (Eq. 1). The values vary from 0.006 to 0.061 $\mathrm{Mg\,ha\,h\,ha^{-1}\,MJ^{-1}\,mm^{-1}}$, and the prediction map shows the highest values for lucerne and the olive orchard, especially where the soils have more silt and VFS, along with less OM and N (see HJ-Biplot). In the surrounding area of the reservoir, the types of soil differ with the topography and land use; therefore, knowledge of soil properties is fundamental when facing the intensification of cultivation that could increase the K factor. These intensive practices decrease OM in soils, making them poor and vulnerable to the soil erosion process.

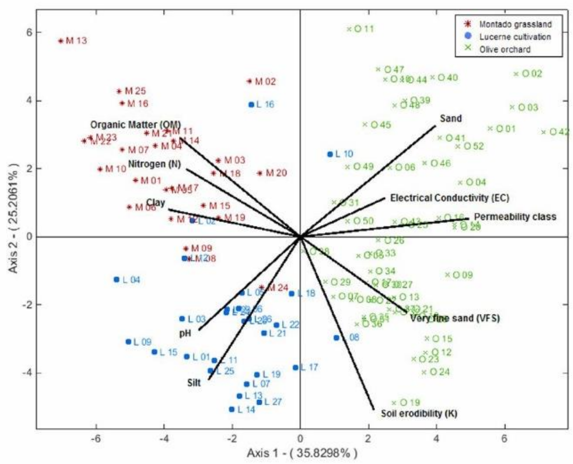

Figure 4. The HJ-Biplot representation matrix of soil samples and studied variables.

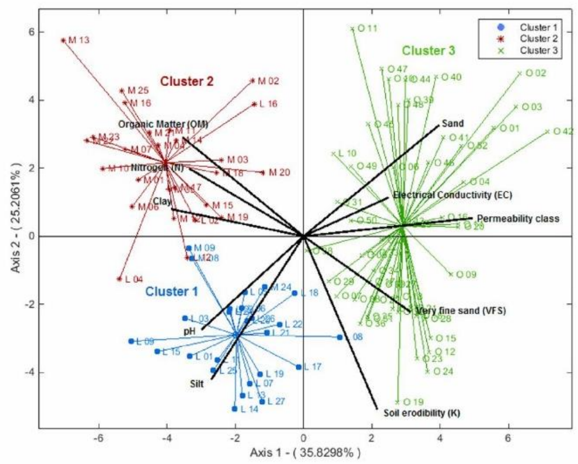

Figure 5. Hierarchical clusters representation of soil samples and studied variables.

When looking for natural vs. anthropogenic impact on the *K* factor, for each land use, it is evident that in the montado the spatial variability is mainly associated with natural (intrinsic) factors (as texture), with soil properties and erodibility distribution being more homogenous. In the lucerne and olive orchard the spatial variability is more dependent on inhomogeneous anthropogenic causes such as fertilization and irrigation rates and tillage/plough processes.

3.4 HJ-Biplot

The HJ-Biplot representation matrix of soil properties is showed in Fig. 4. It was observed that the dominant axis (axis 1) takes 35.83 % of the total inertia (information) of the system. With both dimensions, an accumulative inertia of 61.04 % was achieved. Regarding this graphic representation, it was observed that samples were grouped according to the land use. The montado samples were close to OM, N and clay vectors, showing their preponderance to be a characterization of these variables. The lucerne samples were important to describe the pH and silt content. On the other hand the olive samples were more disperse but related to EC, permeability class, sand, VFS and K.

The variables demonstrating a more positive correlation were OM and N, as previously noticed. Clay and silt were also positively correlated, but they were negatively correlated with sand, as expected, because soils with more sand have less clay and/or silt.

Through the matrix representation it was detected that soils with more sand have higher EC (olive orchard), although EC normally increases with the percentage of clay. This may be explained by the addition of fertilizers, as previously discussed, that can contribute to an EC increase. These results for EC show low variability between land uses, revealing a low cation exchange capacity of these soils. This is frequently caused by intensive soil mobilization (Paz-Gonzalez et al., 2000).

Permeability class increases as the HC_{sat} decreases, as defined by Renard et al. (1997). So, contrary to what was expected, for this study the soils with more sand (occurring in the olive orchard) have less hydraulic conductivity (high-permeability class). It can be explained by a clay-enriched sub-layer under the sandy loam layer or/and by the soil compaction/degradation processes. The soil compaction and degradation can be related to repeated plow operations to reduce shrubs between olive rows and irrigation (Pagliai et al., 2004). This permeability decrease in the olive orchard was correlated with the increase of the *K* factor.

Nevertheless, the properties more positively correlated with the *K* factor were the VFS and silt; this is due to the susceptibility of these particles to erosion since they can be easily detached and transported by water (Morgan, 2005). The OM and N content were negatively correlated with *K* and permeability. The higher OM reduces the susceptibility of the soil to detachment and increases infiltration (Bronick and Lal, 2005). The N content is not used to estimate K; however, especially for soils without fertilization, the existent N is mostly associated with OM. Nevertheless, nutrients decrease in soils that are more erodible, according to the literature (Tesfahunegn et al., 2011). The clay content also shows a negative correlation with *K* factor, as expected (Renard et al., 1997).

Figure 5 shows the hierarchical cluster representation. Using HJ-Biplot methodology and the aggregation tool *ward*, three clusters were obtained. The samples were grouped by land uses (that were already detected by the matrix representation; see Fig. 4). *Cluster 1* is represented by a majority of samples from lucerne, *Cluster 2* by samples from montado and *Cluster 3* by samples from the olive orchard. This was explained by the effect of different management practices, vegetation cover and local soil characteristics, as discussed. Some samples in each land use had different values (higher or lower than the majority) and were grouped in a different *cluster*. Identifying the location of the sample, the cause of displacement can be studied and can help to improve land management practices.

Therefore, the cluster analysis is convenient to identify the effect of different land use and management on soil properties and consequently on soil erosion. On the other hand, the cluster analysis could support the delineation of zones according to soil properties, and subsequently according to erosion susceptibility, which could be used for site-specific soil management recommendations.

4 Conclusions

This study demonstrated that the variability of soil properties and the K factor is associated with land use, cultural practices (tillage type, fertilizer rates, conservation measures, etc.) and local conditions (complex topographic landscape, soil type, etc.). The K factor showed high correlation especially with organic matter, nitrogen, silt and very fine sand. Soils with intensively cultivated land use, and consequently with more tillage and irrigation, had lower organic matter and lower nitrogen content. This translates into a lower cation exchange capacity producing lower aggregate stability and, consequently, an increase of the K factor.

Therefore, in the surrounding area of the Alqueva reservoir, the ongoing change in land use and soil management practices can have a significant effect on chemical and physical soil properties. As a result, this affects the soil erodibility index, intensifying the risk of erosion. The increase of soil loss in the watershed might have a significant impact on a reservoir's ability to store water, reducing its lifespan.

Knowledge of soil spatial variability is fundamental for environment management and can help in the sustainable use of the resource soil. The prediction maps produced with geostatistics are an important monitoring tool, showing the exact position in the field of the specific soil properties. The HJ-Biplot methodology was demonstrated to be useful in gaining a better understanding of how soils properties were correlated and allowed not only a determination of the behavior by sample but also a conclusion as to which variable is responsible for such behavior. The simultaneous use of HJ-Biplot with geostatistics allows this information to be found on the map, which has important theoretical and practical significance for precision agriculture. Facing the intensification of cultivation in the surrounding area of the reservoir, site-specific soil management and careful land use planning are needed to take into account the spatial variability of soil properties, delineating management zones, variable fertilization management, irrigation scheduling, conservation practices and other efforts.

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Appendix V

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Predicting soil erosion after land use changes for irrigating agriculture in a large reservoir of southern Portugal

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Abstract

The construction of Alqueva reservoir in this semi-arid landscape bring new opportunities for irrigated farming. These land use changes may alter the risk of soil erosion that was not predicted in the initial development plans and decrease the lifetime of the investment. A comprehensive methodology that integrates Revised Universal Soil Loss Equation (RUSLE) and Geographic Information Systems (GIS) was adopted to study the effect on soil erosion of different land-uses, taking place on the Alqueva reservoir region. Analysing the soil erosion vulnerability of each land-use ignoring local conditions (not affected by land use), it was obtained: olive orchard>vineyard>montado>lucerne. The strong erosion variances that were observed in the study area show the importance of locating the 'hot spots' of soil erosion. The simulated scenarios can be used as a basis for setting up a Decision Support System that allows the definition of site-specific and restrictive regulations, the development of conservation plans, or provide aid to land owners through support practices and localized environmentally friendly farming.

Keywords: Land-use, soil erosion, seasonal variations, RUSLE.

1. Introduction

Soil erosion by water is one of the most severe and dynamic environmental and economic threats around the world, particularly in regions with seasonal climate and a long past of anthropogenic pressure (Garcia-Ruiz et al., 2013). The intensification of this problem is being frequently associated with land-use changes (Kosmas et al., 1997; Bakker et al., 2008; Leh et al., 2013). Soil erosion decreases the productivity of natural and agricultural ecosystem, since the increase of runoff causes the loss of soil depth, reducing the water and nutrients storage capacity, and thus crop yields (Pimentel, 2006; Li et al., 2009; Hancock et al., 2015). Moreover there is off-site negative impacts associated with the

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increase of runoff that can transport sediments into rivers and reservoirs, causing their pollution and reducing their lifetime (Boardman et al., 2003; Pandey et al., 2007; Ludwig et al., 2009).

The rhythm of Mediterranean land-use changes in the last century has evidently increased, as a consequence of the combination of environmental, economic and social factors (Bakker et al., 2008; Garcia-Ruiz et al. 2013). The linking between different land-uses and soil erosion has attracted the interest of a wide variety of researchers (Kosmas et al., 1997; Cerdan et al., 2010; Blavet et al., 2009; Cerdà et al., 2009; Nunes et al., 2011) who has demonstrated that different vegetation cover and/or agriculture procedures, have different impacts on soil properties and overland flow). Erskine et al. (2002) has demonstrated that land use is the dominant factor determining sediment yields in reservoirs and usually the highest values are associated to the cultivated lands (García-Ruiz and Lana-Renault, 2011).

Over the past few decades, numerous advances have been made to assess soil erosion, to overcome the costs and unfeasibility of monitoring in situ. So, there has been substantial investigation into soil loss models that vary according complexity, processes accounted and the information required (Merrit et al., 2003; Bhattarai and Dutta 2008; Volk et al. 2010). These models have been shown to be useful in combination with Geographic Information Systems (GIS) techniques and remote sensing because it allows the spatial distribution of soil erosion processes with reasonable expenses and accuracy (Terranova et al., 2009; Prasannakumar et al., 2011). The empirical Universal Soil Loss Equation (USLE) is one of the most widely used models for estimating annual soil loss, (Wischemeier and Smith 1978) from agricultural watersheds and its modifications include the Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1997). Although some disadvantage reported (Volk et al., 2010), RUSLE is easy and valuable to use, because of the structure simplicity, low input data requests and the availability of parameter values.

The Alqueva reservoir surrounding, a Mediterranean region, has faced new challenges and some land-use changes have been happening, as a consequence of water availability. The intensification of irrigated farming has been occurring simultaneously with the abandonment of the typical agroforestry system, increasing the need to promote the sustainability, decreasing soil erosion and protecting the reservoir. The main objective was to study the effect of typical and new land-uses on soil erosion, accounting for seasonal variations on rainfall and vegetation cover during the year.

2. Study area

2.1. The Alqueva dam region

The Alqueva reservoir is located on the Guadiana river in the south of Portugal (8º30' W, 38º30' N) (Fig. 1).

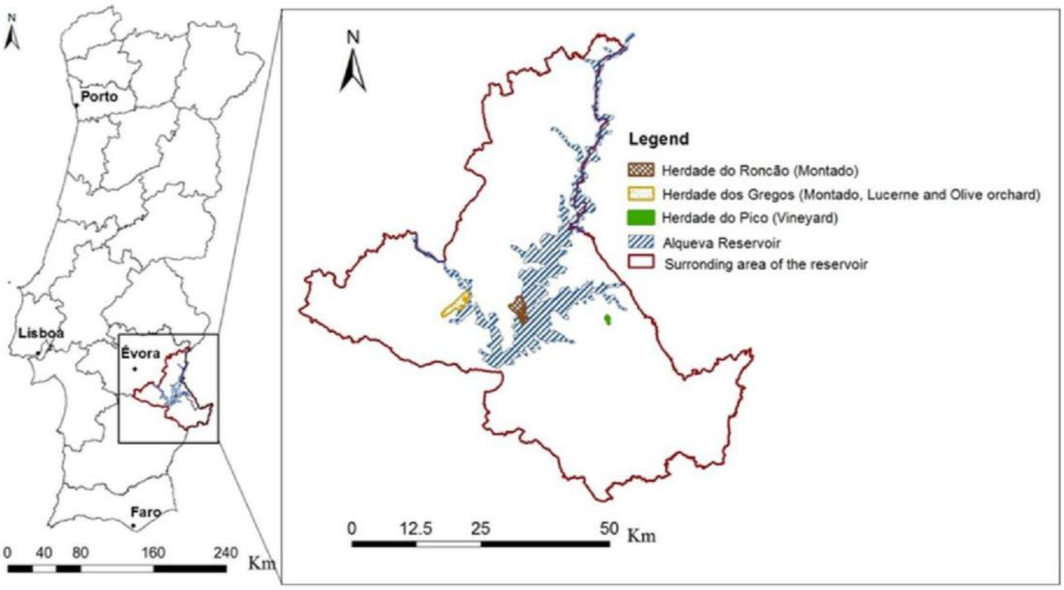

Fig.1 - Location of the Alqueva reservoir and experimental study areas.

The reservoir covers an area of 250 km^2 (from which 35 km^2 are in Spain) and the total capacity is 4150 hm^3. The lake total shoreline is approximately 1100 km, it extends for 83 km and is considered one of the biggest in Europe (Lindim et al., 2011). The Alueva project was constructed during 1998-2002 (Fig. 2a), and the main objective was to create a strategic water reserve for population consumption, for agriculture irrigation (about 110000 ha), for energy production, as well as for landscape enhancement where several tourist projects can be built. Alqueva dam has direct influence in the regions surrounding it (namely 18 counties).

This region of Alentejo, Southern Portugal, is characterized by a complex landscape structure. A traditional landscape is named "Montado", an agroforestry system in which agricultural and forest activities complement each other comprising an open formation of oak species combined with a rotation of crops/pastures. There was an intensification of agriculture (cereal production) in combination with extensive livestock breeding in the beginning of the 20th century, which lead for numerous environmental impacts and namely increased soil erosion. Though, especially since Portugal joined the European Community in 1986, the abandonment of agricultural activities in Alentejo increased, and the "montado" system is in transition towards a silvo-pastoral or even purely forestry system (Pinto-Correia e Mascarenhas, 1999). Today, with Alqueva reservoir, the intensification of some farming systems occurs simultaneously with the reduction and ultimate abandonment of others.

The climate is Mediterranean, with hot and dry summers and mild winters. The annual mean for temperature ranges from 24 to 28 °C in hot months (July/August), and from 8 to 11 °C in cold months (December/January). The annual mean of precipitation ranges between 450 and 550 mm, however the region is affected by intense dry periods without precipitation, since almost 80% of the precipitation occurs from October to April.

2.2. Experimental study areas

In order to study the effect of land-use on soil erosion we identified three areas (called in Portuguese "herdades") namely: "Herdade do Roncão", "Herdade dos Gregos" and "Herdade do Pico". Four different land-uses were identified and selected to study namely: montado grassland, vineyard, lucerne cultivation, and olive orchard (Fig. 2).

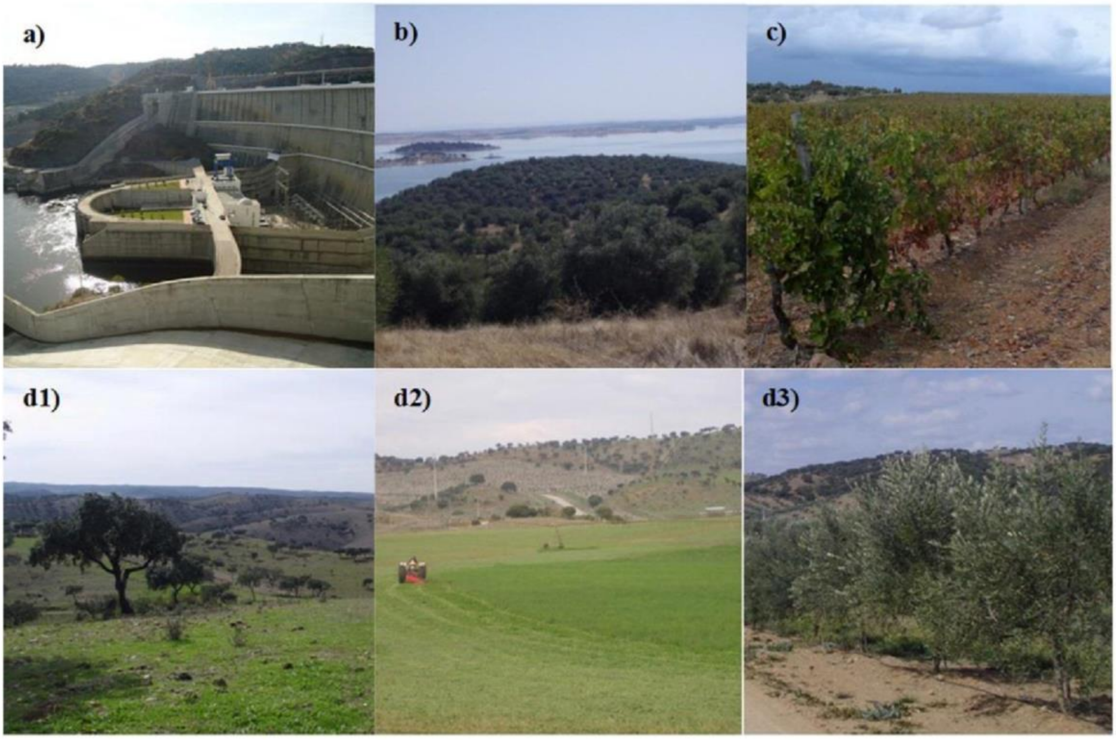

Fig. 2 - Alqueva dam project and land-uses (experimental sites): a) Alqueva dam; b) Montado in the "Herdade do Roncão"; c) Vineyard in the "Herdade do Pico"; d1) Montado, d2) Lucerne and d3) Olive orchard in the "Herdade dos Gregos".

The "Herdade do Roncão" (739 ha) includes one land-use:

- A typical "Montado" (Fig. 2b), characterized by a silvo-pastoral system with low density holm oaks and some olive trees, where farming was abandoned for about 6 years ago.

The "Herdade do Pico" (30 ha) consists also of a one land-use:

- A vineyard with irrigation (Fig. 2c), plough between lines and fertilization.

The "Herdade dos Gregos" (900) comprises three different land-uses studied:

- A Montado (Fig. 2d1) similarly characterized by a silvo-pastoral system with low density holm oaks and which suffered a fire 3 years ago;
- An intensive cultivation of Lucerne (Fig. 2d2) with irrigation, tillage and fertilization;
- An olive orchard (Fig. 2d3) with irrigation system, plough between lines and fertilization.

3. Methodology

This study integrated the use of RUSLE equation, GIS, remote sensing and geostatistics to predict soil erosion and respective factors. RUSLE, is defined as:

$$A=R\ K\ L\ S\ C\ P \tag{1}$$

where A= potential annual erosion (t ha^{-1} year^{-1}), R= rainfall and runoff erosivity factor, K= soil erodibility factor, LS= slope length and gradient factor, C= vegetation cover factor and P= support practice factor (Renard et al., 1997). To study seasonal soil erosion, the erodibility (K) and topographic (LS) factor were aggregate as "static" over the year, and rainfall erosivity (R) and vegetation cover (C) factors were analysed per season. The nex sections describe the RUSLE data collecting and processing for each factor. The spatial data was easily treated with ArcGIS software.

3.1. Soil erodibility factor (K)

Soil erodibility factor (K) represents the susceptibility of a soil to erode and the amount and rate of runoff, as measured under continuously cultivated fallow plot, 72.6 ft (22.1 m) long with a slope of 9% (Renard et al., 1997). It is a quantitative value experimentally determined using an algebraic approximation (Wischmeier and Smith, 1978):

$$K = [2.1 \times 10\text{– }4(12\text{–}OM) \times M1.14 + 3.25(S - 2) + 2.5(P - 3)]/100 \tag{2}$$

where OM is organic matter, s is soil structure, and p is permeability class. M is the product of the primary particle size fractions (% MSilt) × (%MSilt + %MSand), where %MSilt is percent modified silt (0.002-0.1 mm), and % MSand is percent modified sand (0.1-2 mm). Modified silt is the amount of silt particles and very fine sand, considered the most susceptible particles to erosion, because can be easily removed by the raindrop splash and runoff water.

A minimum of 25 soil samples with 20 cm depth were collected in each area. The sample localizations, in field, were saved using a Global Positioning System (GPS). Soil permeability and structure was estimated in the field. In the laboratory, the particle-size distribution, soil organic matter (OM) and soil total nitrogen (N) were investigated.

The data were subjected to statistical analysis using SPSS 17.0 software, namely the mean, standard deviation (SD), minimum and maximum, coefficient of variation (CV) and skewness of each parameter. To evaluate significant differences between means for each land-use we used multiple ANOVA (Duncan multiple range test). A continuous surface representing the spatial variation of this factor was prepared in ArcGIS 10 software using geostatistics tool.

3.2. Slope length and steepness factor (LS)

Slope Length (L) and slope steepness (S) factors show the influence of topography on soil erosion erosion (Wischemier and Smith, 1978). Direct measurements of slope and slope length were initially proposed to evaluate these factors (Renard et al., 1997). However this method is only suitable for small plots and parcels, because intensive field measurements are obviously not feasible on a regional scale. In watershed scale, the use of a Digital Elevation Model (DEM) in GIS, for data input is a better approach (Nekhay et al., 2009). Therefore, in the present study, a DEM from the region was used in ArcGIS software to estimate this factor. The combined LS factor (without units) was computed using ArcGIS spatial analyst extension, following the equation (3) (Moore and Wilson, 1992):

$$LS=(\text{flow accumullation} \times \text{cell size}/22.13)^{p} (\sin \alpha/0.0896)^{q} \quad (3)$$

where p and q are empirical exponents (p = 0.4 and q = 1.3) (Moore and Wilson, 1992), flow accumulation signifies the accumulated upslope contributing area for a given cell, cell size is the size of DEM grid cell and α is the slope degree value.

3.3. *Rainfall erosivity (R)*

The rainfall-runoff erosivity (R) represents the erosive potential of raindrops impact and of runoff generated by erosive storms. According to Renard et al. (1997), the rainfall-runoff factor is determined through the sum of erosive storm values EI30 occurring during a mean year, which result for the product of total storm kinetic energy (E) times the maximum 30 minute intensity (I30), where E is in MJ/ha and I30 is in mm/h. The monthly erosive storm empirical index EI30 was computed for 25 meteorological stations in the surrounding area of the Alqueva using the regression equation (Goovaerts, 1999):

$$EI30month=6,56 \times rain10-75,09 \times days10 \quad (4)$$

where rain10 is the monthly rainfall for days with rainfall higher than 10 mm and days10 the monthly number of days where rainfall exceeds 10 mm. Daily precipitation data during 30 years (1980-2010) were used. Rainfall erosivity maps per season, for the region, were created using geostatistics. The R means for each land-use were estimated.

3.4. *Cover management factor (C)*

The C factor reflects the effect of vegetation on erosion rate (Renard et al., 1997), considering that it reduces the erosive impact of rainfall and slow down overland flow. C factor values diverge from 0 (well-protected soil) to 1 (bare soil) and there is a strict relation with land use types. Remote-sensing has been one of the most widely used methods for mapping the C factor (Van der Knijff et al., 1999; Prasannakumar et al., 2011), because vegetation cover can be estimated using vegetation indices derived from satellite images such as Normalized Difference Vegetation Index (NDVI). NDVI is an indicator of vegetation growth and ranges from -1 to 1. This method gives different prospective on soil erosion studies because allows the estimation of intra-annual changes in vegetation through images for different periods (Ouyang et al., 2010). NDVI was computed utilizing band 3 (red) and band 4 (near-infrared) as follows:

$$NDVI= (Band4 - Band3)/(Band4-Band3) \quad (5)$$

Satellite images from different seasons with a spatial resolution of 30 m were used. To estimate C factor, the most common procedure using NDVI involves the use of regression equation model derived from the correlation analysis between the C factor values measured in the field and a satellite-derived NDVI. Landsat TM images were processed to obtain NDVI and the equation was used to generate a C factor (Van der Knijff et al., 1999):

$$C= e^{(-\alpha(NDVI/(\beta-NDVI))} \quad (6)$$

where α and β are unitless values that determine the shape of the curve relating NDVI and C factor. Van der Knijff et al. (1999) found that this approach gave better results than assuming a linear relationship, and the values of 2 and 1 were selected for the parameters α and β, respectively. C factor maps were produced with ArcGIS software.

3.5. *Conservation practice factor (P)*

The support practices factor (P) reflects the effects of specific practices that can be used to reduce the amount and rate of erosion, such as contouring, strip-cropping, terracing, and subsurface drainage. These practices affect erosion by modifying the flow pattern, grade, or direction of surface runoff and by reducing the amount and rate of runoff (Renard et al., 1997). In this study, P factor was assigned the value of 1 (no support practice factor) for all land-uses, because the support practices in these areas are not relevant or not existent.

4. Results and Discussion

To better understand the effect of each land-use on soil erosion we examined each factor considered by RUSLE.

4.1. Soil erodibility (K)

Soil erodibility was considered time invariant. The mean values of some soil properties and estimated soil erodibility for each land-use are presented in Table 1, as well as Duncan test results. Duncan's test showed significant differences between land-uses for all properties analysed. Means with different letters in the same property are significantly different between land-uses at $p \leq 0.05$. It reveals that land-uses have highly significant effect on soil properties and in turn on soil erodibility.

Table 1 – Soil properties and soil erodibility (K) means.

	H. Roncão	H. Gregos			H. Pico
	Montado	**Montado**	**Lucerne**	**Olive Orch.**	**Vineyard**
Sand (%)	62.71b	53.16a	52.93a	65.81b	56.5a
Silt (%)	22.01a	29.55b	33.79c	24.37a	21.8a
Clay (%)	15.27b	17.29c	13.29b	9.83a	21.6d
OM (%)	4.63c	5.22c	2.08b	2.10b	0.77a
N (%)	-	0.19c	0.11b	0.10b	0.07a
K factor (t ha h ha-1 MJ-1 mm-1)	0.023a	0.021a	0.039c	0.038c	0.029b

Means with different letters in the same property are significantly different between land-uses at $\mathbf{p} \leq 0.05$ (Duncan's test).

Despite some significant differences, from the particle size distribution, the soils are all sandy loam, formed mainly with sand, following by silt and low amounts of clay. Though, it's clearly that the content of organic matter (OM) and nitrogen (N) decrease with the intensification of land-use (lucerne cultivation, olive orchard and vineyard) and the differences are significant. The low values of OM and N in those intensive land-uses can be explained with the soil mobilizations and irrigation system. Soil tillage mixes the subsoil with topsoil, thus there is more OM decomposition and water erosion easily removed the nutrients from surface (Tesfahunegn et al., 2011).

As a result, soil erodibility increased with the intensification of cultivation field, with lowest values for Montado grassland (0.021 t ha h ha^{-1} MJ^{-1} mm^{-1}) and highest for Lucerne (0.039 t ha h ha^{-1} MJ^{-1} mm^{-1}). Other studies had similar results, showing that the removal of permanent vegetation, the loss of OM and the reduction of aggregation, caused by intensive cultivation, contribute to decrease of soil erodibility (Evrendilek et al., 2004).

4.2. Topography (LS)

The LS factor maps for each area are presented in the Figure 2. The highest LS factor values occur for Montado areas (mean of 1.28 and 1.86 for "Herdade do Roncão" and "Herdade dos Gregos" respectively) and for vineyard (mean of 1.21), and are mainly associated to great slopes. Through the prediction LS factor maps its easy the identification of sensitive areas for some land-uses, because it has been demonstrated that increases in this factor can produce higher overland flow velocities and correspondingly higher erosion (Van Remortel, 2004).

4.3. Soil erosivity (R)

Rainfall erosivity values were estimated for 25 stations and using geostatistic techniques we obtained a regional erosivity map for each season. The mean values of soil erosivity were estimated for each experimental area and are shown in Table 2. As shown in table, the values vary lightly between areas despite approximate locations, and these variances were taken into account when predicting soil erosion. Looking at values for each season we noticed that rainfall erosivity is characterized by a strong seasonality and the highest rainfall erosivity values are related with the first rainfall events after summer, in autumn, and the lowest values occur in summer. In percentages, about 47-50% of annual rainfall erosivity occurs in autumn, 20-24% in winter, 17-19% in spring and only 9-10% in summer. These trends are comparable to the results found in other Mediterranean studies (Van der Knijff et al., 1999, Diodato, 2004).

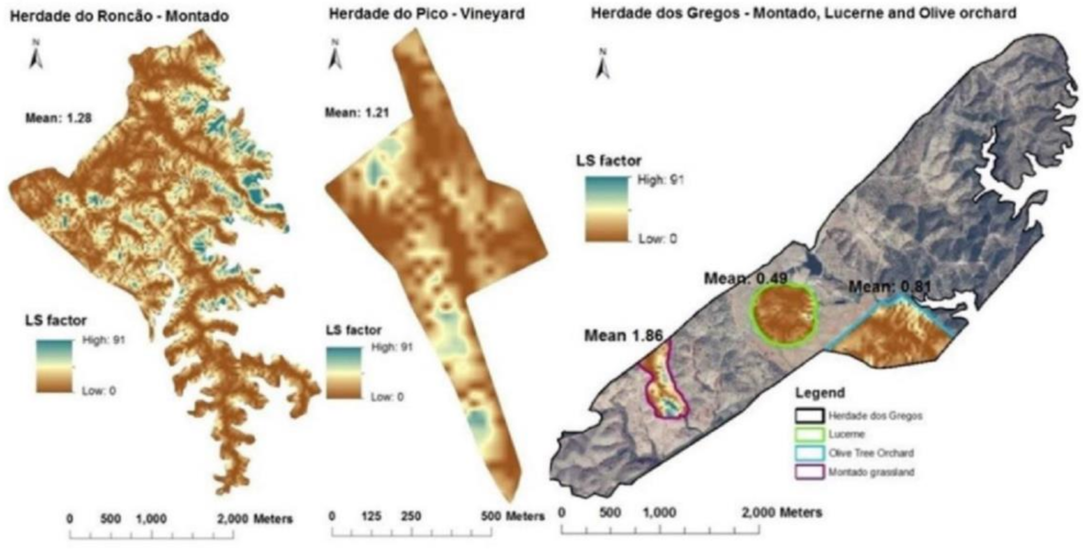

Figure 1 – LS factor for each study area.

Table 2 – Mean values of rainfall erosivity for all experimental study areas, according each season.

	H. Roncão	H. Gregos	H. Pico
Autumn	423.8	445.7	404.1
Winter	193.4	212.2	170.0
Spring	146.4	152.6	158.7
Summer	79.7	79.9	81.2
Total	875.6	892.0	851.7

4.4. Vegetation Cover (C)

The NDVI values got from satellite and the calculated C values, accounting seasonality, are shown on Table 3.

Table 3 – Mean values of NDVI and C factor for each land-use.

		H. Roncão	H. Gregos			H. Pico
		Montado	Montado	Lucerne	Olive Orc.	Vineyard
Autumn	NDVI	0.110	0.127	0.673	0.156	0.152
	C factor	0.789	0.747	0.035	0.690	0.697
Winter	NDVI	0.455	0.494	0.581	0.513	0.193
	C factor	0.193	0.141	0.065	0.125	0.620
Spring	NDVI	0.222	0.241	0.226	0.169	0.224
	C factor	0.567	0.531	0.558	0.665	0.562
Summer	NDVI	0.102	0.107	0.138	0.110	0.138
	C factor	0.797	0.790	0.724	0.780	0.726

As observed the NDVI and vegetation cover factor (C) are negatively correlation, since higher NDVI values means greater vegetation cover that implies lower soil erosion (lower C factor). So, the highest NDVI values were found to be higher principally during the winter and the lower during the summer, except for lucerne and vineyard. For Montados and Olive orchard the vegetation cover is more seasonal dependent, being the highest vegetation cover during the winter (C factor lower than 0.2) and the lowest in summer and autumn (C factor greater than 0.69). On the other hand in lucerne cultivation and vineyard it's more reliant on of farming practices. For lucerne land-use the highest vegetation cover (NDVI) occurs also during winter but especially in autumn possible due to irrigation system (C factor lower than 0.1). The vineyard has the low vegetation cover during the all year, including winter (C factor higher than 0.6), explained by soil mobilizations to remove natural vegetation between lines.

4.5. Soil erosion

All factors were integrated in ArcGIS spatial analyst to obtain and quantify soil erosion rates for each season and annually. To easily compare values between different land-use the estimated means of soil erosion are shown in Table 4. For almost land-use (except to Lucerne), the soil erosion is expected to be higher during autumn season (more than 8 t/ha), contributing with more than 50% of annual vegetation. In this period the soil erosivity reaches the peak due to heavy rainstorms and the soil is very susceptible since vegetation cover is still low-moderate after the summer, the dry season with high temperatures. Despite higher soil erodibility (K) for Lucerne, the annual erosion is the lowest resulting from low slopes (LS) and vegetation cover (C) especially during the season with more rainfall erosivity. On the other hand, the Montado in the "Herdade dos Gregos" show the maximum soil erosion, despite lower erodibility (K), a consequence of greater slopes (LS) and low vegetation since it suffered a fire 3 years ago.

Table 3 - Soil erosion mean values for each land-use and each season.

	H. Roncão	H. Gregos			H. Pico
	Montado	Montado	Lucerne	Olive Orc.	Vineyard
Autumn	9.730	12.770	0.447	9.438	8.823
Winter	1.031	1.175	0.303	0.826	3.440
Spring	2.438	2.736	1.615	3.068	2.778
Summer	1.820	2.267	1.117	1.908	1.851
Anual	15.036	19.108	3.502	15.232	16.892

The RUSLE factors were integrated and were generated seasonal annual prediction maps of soil loss shown in Figure 5. The maps analysis allows the identification of sensitive areas and it is evident the topographic influence on soil erosion rates. Throughout the prediction maps it can be seen easily the seasonal variations.

The topography (LS) and the rainfall erosivity (R) are characteristics of local not influenced by land-uses, considering mainly factors that differ with land-use area vegetation cover (C) and the soil erodibility (K). Therefore, a lastly analysis was done to compare the effect of land use, ignoring R and LS factores. Considering only the KC ratio we obtained the following erosion susceptibility: Olive orchard>vineyard>Montado>Lucerne. Similar results were obtained by different authors, which observed higher soil erosion rates in orchards and vineyards comparatively to woodlands, scrubland or fire affected land (Cerdá *et al.*, 2009, Kosmas *et al.*, 1997).

5. Conclusions

The great soil erosion rates variability reflects the importance of studying different scenarios of land-use, accounting for seasonal variations of vegetation and rainfall. There are important differences in hydrological functioning and erosional response of soils under multiple land uses and vegetation types.

Since the intensive irrigated cultivation is increasing in the region, which tends to affect susceptibility of soil to erode, it is important to ensure the vegetation cover mainly during the periods with intensive rainfalls and in the sensitive areas. In the olive orchard and vineyards, alternatively of periodic vegetation removal, keep vegetation growing between lines is an example of conservation technique, usually called "strip cropping". The vegetation cover in lucerne cultivation proves to protect soil during intensive rainfall periods, though it is important to understand soil degradation caused by intensive cultivation, tillage and fertilization. The abandonment of montado systems leads to poorly managed forest areas and constrains the sustainability of the system, increasing forest fires risk and soil erosion.

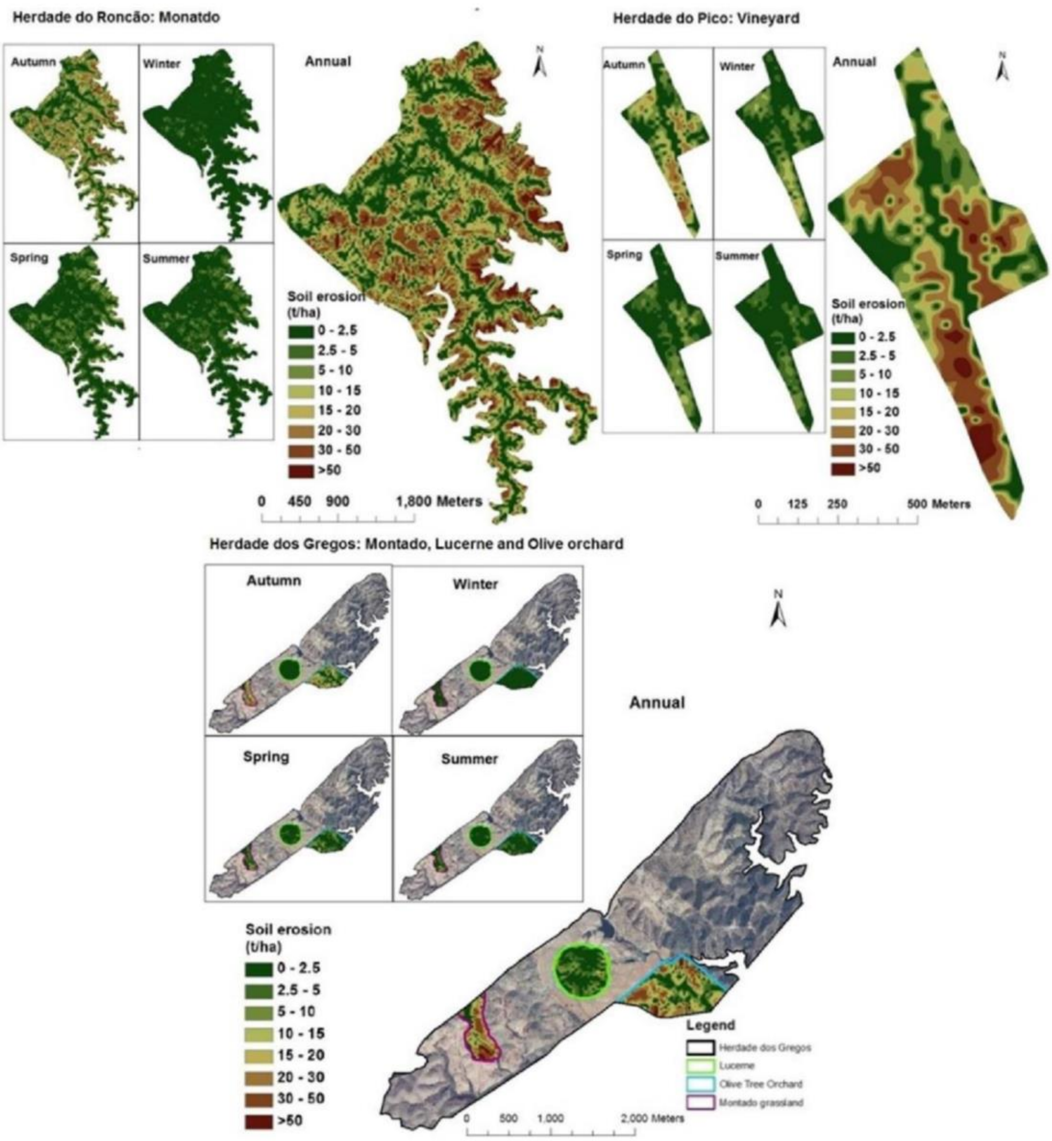

Figure 4 – Prediction maps of soil erosion for each land-use, accounting seasonal variations.

The soil erosion prediction maps for different scenarios and the contribution of each factor will be used as a solid base to create a Decision Support System (DSS). This should promote sustainable management of different land-uses in the region, based on spatial variability of soil characteristics and topography, and on seasonal variations. Site specific methods and mitigation measures should allow soil erosion reduction and consequently protect reservoir.

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