A LANDUSE CHANGE AND LAND DEGRADATION STUDY IN SPAIN AND GREECE USING REMOTE SENSING AND GIS

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Commission VII, WG VII/4

KEY WORDS: Remote Sensing, GIS, Land Cover, Land Use, Change Detection, Classification, Hydrology, Soil

ABSTRACT:

The relationship between landuse/landcover (LULC) changes and land degradation in two Mediterranean sites is investigated using remotely sensed and ancillary data. The areas of study, the Xaló river catchment in the north of the Alicante province in southeast Spain and the Aegean island of Lesbos, Greece, have both been subjected to changes in LULC, such as abandonment, overgrazing, forest fires and tourist development. Landsat MSS data dating back to the 1970s were used for the mapping of historic landuse/cover types whereas Landsat TM and ETM+ data were employed for the analysis of their recent state. A soil erosion model was then used within a GIS in order to study the susceptibility of the areas affected by changes to overland flow and rainsplash erosion. The model consists of four parameters, namely soil erodibility, slope, vegetation cover and overland flow. The results show increased susceptibility to runoff and erosion mostly for those areas were forest fires, urbanization, and/or overgrazing were the main causes of change and suggest that mitigation measures should be taken for the prevention of further degradation. The readily implemented methodology proposed, based on modest data requirements, is a useful tool for catchment to regional scale LULC change and land degradation studies.

1. INTRODUCTION

Over the last ten years, a lot of attention has been drawn on the issue of landuse and landcover (LULC) changes and the direct or indirect relationship these changes might have with the observed land degradation in the Mediterranean region (Brandt and Thornes, 1996; Drake and Vafeidis, 2004; Thornes, 1996). Such changes are the result of practices such as the relocation of people to the coastal border, farm and grazing abandonment inland, the explosion of tourism-related activities, and the intensification of agriculture, among others. Accurate LULC mapping over large areas has become necessary in order to monitor these changes and has received a considerable boost from the advent of multispectral satellite data. Such data have become operationally available since the early 1970s and have paved the way for LULC and vegetation cover studies due to their suitable spectral, spatial and temporal resolution, thus providing scientists with a useful tool to study LULC changes and their relationship with land degradation processes.

There is a well-established tendency for water runoff to increase with land degradation (Symeonakis and Drake, 2004; UNEP, 1990; Rubio and Bochet, 1998; Krugmann, 1996; Sharma, 1996; Kosmas et al. 1999). Overgrazing, for example, leads to trampling and compaction of the soil which reduces the infiltration and thus increases the amount that leaves as runoff. Deforestation also leads to increased overland flow since it removes the vegetation which probably affects rates of runoff more than any other single factor. The rate of runoff is therefore a useful indicator of the land degradation and desertification process and was estimated in the present study with the use of the Soil Conservation Service (SCS) model (SCS, 1972).

The importance of soil erosion in land degradation and desertification over the Mediterranean area has also been widely recognized since it appears to be the end result of almost all such processes (Brandt and Thornes, 1996; Kosmas et al., 1999; Symeonakis and Drake, 2004). Drought, the natural or human induced reduction in vegetation cover, poor agricultural practices leading to soil aggregate breakdown and soil organic matter losses, poor irrigation practices leading to salinisation, all lead to an increase in soil erosion rates and ultimately desertification. Erosion therefore seems to be the single most important indicator of the land degradation and desertification processes and was estimated here with the use of the Thornes model (Thornes 1985, 1989).

The main aim of this research was therefore to study the interrelationship between LULC change and land degradation over two different Mediterranean sites using remotely-sensed data. Specific objectives included the investigation of the feasibility of the combination of remotely-sensed data in land degradation studies, the estimation of LULC changes, overland flow and sheetwash soil erosion, and to suggest a methodological framework that provides a tool for the appraisal of the impact of changes in land degradation.

2. THE STUDY AREAS

One of the two study sites is the catchment of the Xaló river in the North of the Alicante province in Southeast Spain (Figure 1). It covers an area of approximately 30200 ha, is characterised by a complex topography which ranges from 0 to 1365m above sea level, diverse microclimatic conditions with irregular and intense rainfalls and soils mainly falling under the Cambisols and Regosols types. As previous studies have shown (Belda 1997, Symeonakis et al., 2003; Symeonakis et al., in press; Viedma 1999, Viedma and Meliá, 1999), the area has been subjected to a number of landuse/cover changes during the 20th century, such as a number of forest fires, tourist development mainly around the coastal rim and, abandonment of agricultural terraces inland.

The other area of study is the northern Aegean island of Lesbos, Greece, which covers an area of approximately 163000ha and has a maximum altitude of 947m. The climate is characterised by strong seasonal and spatial variations of rainfall and high oscillations between minimum and maximum daily temperatures, typical of the Mediterranean region (Kosmas et al., 2000b). Kosmas et al. (2000a) divided the island in three climatic zones: the semi-arid in the West with an average annual rainfall of 415mm, the largest dry sub-humid zone in the East with 677mm and a transitional zone between the two. The soils are developed on various lithological formations such as shale, schist-marble, volcanic lava, pyroclastics, and ignimbrite and are classified as Typic Xerochrept, Lithic Xerochrept, or Lithic Xerorthent (Kosmas et al., 2000a; Kosmas et al., 2000b; Loumou et al., 2000).

3. METHODS AND RESULTS

3.1 Data

The datasets available for the Xaló area were: (a) a Landsat ETM+ image taken in August 2000, (b) a Landsat MSS image taken in July 1978, (d) a Digital Elevation Model (DEM) with a 25×25m resolution, (e) 1:50000 aerial photographs taken in 1977, (f) 1:30000 aerial photographs taken in 1997, (g) soil samples at 16 point locations in the wider catchment area and (h) digital 1:50000 soil erodibility and lithological data. For the island of Lesbos the respective data were: (a) a Landsat TM image taken in August 1999, (b) a Landsat MSS of May 1975, (c) a DEM with a 30×30m resolution, and (d) 1:200000 soil map.

3.2 Landuse changes

3.2.1 LULC changes in Xaló

The mapping of the various types of LULC for the area of Xaló was achieved using the Landsat data and the fuzzy classification and fuzzy convolution modules of Erdas Imagine 8.5. The data were first rectified for the effect of relief using the 25×25m DEM and projected to the right geographical projection using 54 ground control points with a total RMSE of 25.24m. In order to further assist the fuzzy classifier in distinguishing between the different classes, one additional layer of information was stacked to the four MSS bands of the 1978 image and the six ETM+ bands of the 2000 image, namely the Normalised Difference Vegetation Index (NDVI, Justice et al. 1985), thus forming a 5-layered and a 7-layered image, respectively. The four principal components of these multi-layered images were then extracted, the last three of which were then used for the delineation of the sampling areas that would serve as 'training' for the classifier. Five fuzzy layers per pixel were used and the resulting classified images were assessed using 100 random points and the aerial photographs as ground truth. The accuracy of the classification methodology was tested only for the 2000 data and produced an overall accuracy of 86% and an overall kappa statistic of 0.82. These figures compare favourably with the results of a hard maximum-likelihood (ML) classification of the same area (Symeonakis et al., 2003; Symeonakis et al, in press).

The result of the fuzzy classification for 1978 (Figure 1a), shows that the vast majority of the catchment area was covered by the various types of orchards, with an area of approximately 12392ha or 41% of the entire catchment. The second LULC type, in terms of area covered, was matorral (7541ha), followed closely by 'bare' (7363ha), the two of which together cover almost half of the entire catchment area (49%). Finally, forests, urban areas and the various horticultural types all shared approximately a 3% of the entire catchment area, with 1039ha, 889ha and 879ha, respectively. For the year 2000 (Figure 1b) the fuzzy classification gives the following figures: matorral 9480ha (31%), orchards 8955ha (30%), bare 6508 (22%), horticulture 2527ha (8%), urban areas 1937ha (6%), and forests 817ha (3%).

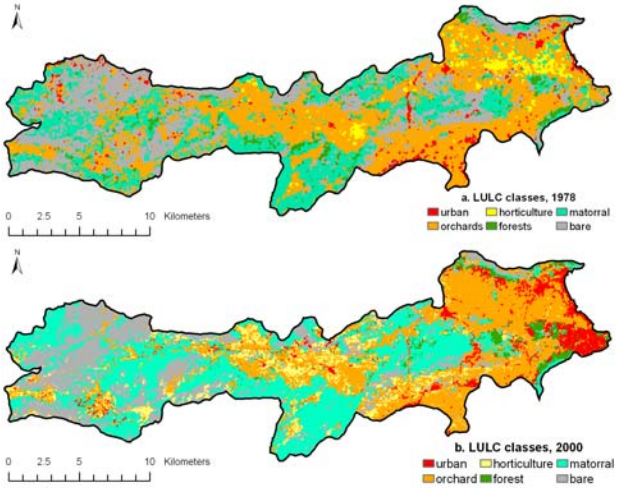

Figure 1: Landuse/landcover (LULC) maps of the Xaló catchment with predominant LULC types produced from fuzzy classification of, (a) Landsat MSS data of 1978, and (b) Landsat ETM+ data of 2000

A comparison of the landuse/landcover maps for 1978 and 2000 in figures 1a and 1b respectively, produced the graph in Figure 2 below:

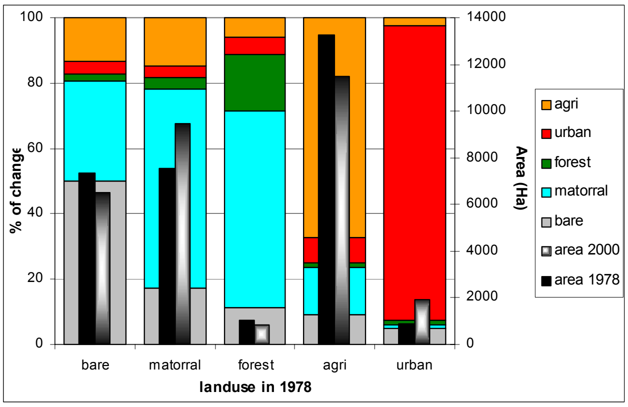

Figure 2: Percentage of change observed between 1978 and 2000 for five distinct types: bare, matorral, forest, agricultural (orchards and horticultural together) and urban, along with the area (ha) occupied by each of the types in both years

Some of the changes that appear to have taken place between 1978 and 2000 have a direct relationship with the land degradation process, namely:

- A 21% decrease in forested areas (223ha), most likely due to the large number of fires that have occurred in the area (Belda 1997).
- A 13% loss of agricultural land to matorral (1890ha) and urban areas (1038ha), due to the abandonment of agricultural terraces.

- A 113% increase in urbanised areas

On the other hand, some of the changes observed in other areas have a positive effect with respect to land degradation, such as:

- A conversion or rehabilitation of 12% of the bare areas in 1978 to matorral (2237ha), agricultural land (994ha) and forests (161ha).
- A further reforestation of former agricultural terraces (209ha) and matorral (258 ha).

3.2.2 LULC changes in Lesbos

The mapping of the various LULC classes over the island of Lesbos was carried out using the Landsat MSS and TM images of 1975 and 1999, respectively. The methodology applied involved a ML classification which was then tested randomly distributed ground truth data acquired with the use of GPS. The overall accuracy was 81% for the 1975 data and 89% for the year 1999. Figure 3a is the resulting LULC map for the year 1975 and 3b is the same for the year 1999. Misregistration errors were not quantified at this stage.

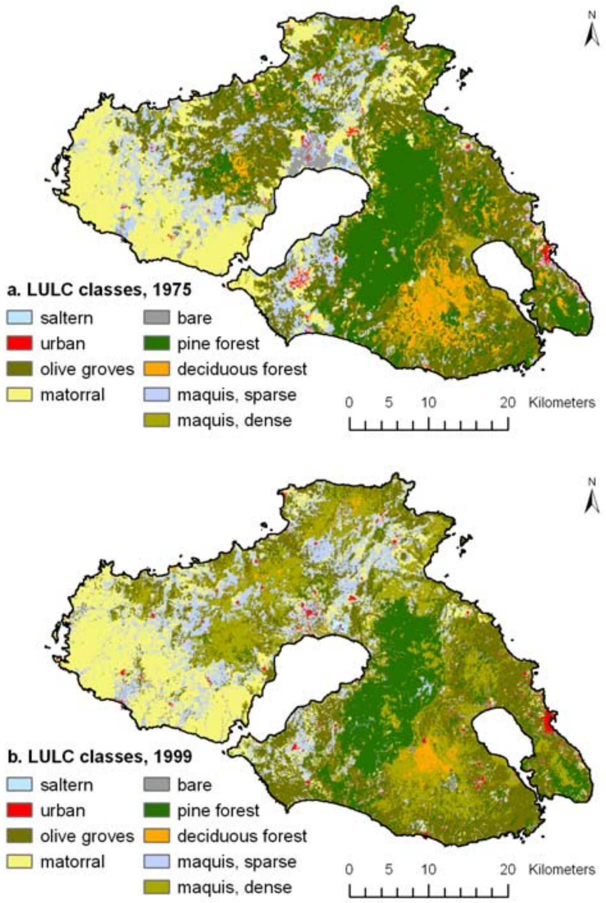

Figure 3: Landuse/landcover (LULC) maps of the island of Lesbos with predominant LULC types produced from ML classification of (a) Landsat MSS data of 1975, and (b) Landsat TM data of 1999

The result of the ML classification for 1975 (Figure 3a), shows that the greatest part of the island was covered by olive groves, covering approximately 30% of the entire island. Matorral and pine forests both covered about 20% of the island area, followed by sparse (14%) and dense maquis (8%), respectively. Deciduous forests covered almost 5% of the total area in 1975, while bare and urban areas occupied a mere 1%. In the year 1999 (Figure 1b), the olive groves were again the predominant landuse, occupying a slightly smaller area than they did in 1975. Dense maquis were now the second largest LULC type with 34133ha or 21% of the total area, followed by matorral (30576ha or 19%), sparse maquis (22530ha or 14%), pine forests (20168ha or 12%), bare (5213ha or 3%), oaks (1797ha or 1%) and urban areas (1399ha or 1%). A comparison of the LULC maps for 1975 and 1999 in figures 3a and 3b respectively, produced the graph in figure 4 below:

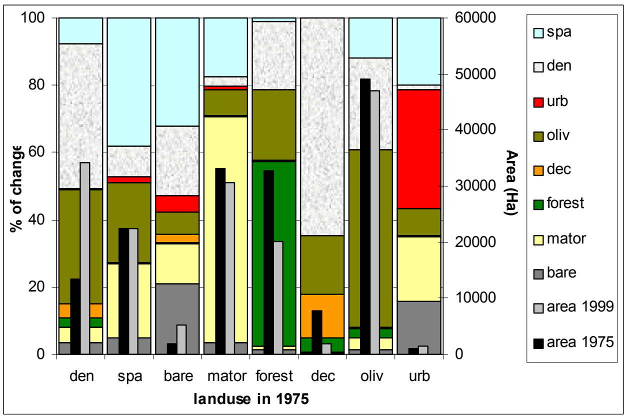

Figure 4: Percentage of change observed between 1975 and 1999 for eight distinct types: bare, matorral, pine forest, deciduous forest, olive groves, sparse maquis, dense maquis and urban, along with the area (ha) occupied by each of the types in both years

Some of the changes that appear to have taken place between 1975 and 1999 are directly related to degradation processes, namely:

- An important decrease in forested areas, due to the large number of fires that took place on the island in the 1980s and the 1990s.
- An increase of bare areas, from 1890ha in 1975 to 5213ha in 1999.
- A 4% loss of olive groves and their conversion to pasture due to their location in inaccessible mountainous areas. Over the last few decades, farmers in Lesbos have partly turned to tourism and tourism related activities while complementing their income from olive cultivations (Loumou et al., 2000). This is why only a relatively small percentage of olive groves were lost.
- The increase in urban land (23%) is partly because of tourism, but mainly takes places in areas near the capital of the island (Mytilene) where the population is increasing.

3.3 Runoff modelling

The SCS Curve Number method (SCS, 1972) was utilised for the estimation of event runoff. The model was developed by studying overland flow in many small experimental catchments and is one of the most widely used methods to compute direct storm runoff (SCS, 1972; Maidment, 1993). The general form of the relation is 'well established by both theory and observation' (Maidment, 1993). No runoff occurs until rainfall (P) equals an initial abstraction I_a. After allowing for I_a, the depth of runoff Q is the residual after subtracting F, the infiltration of water retained in the drainage basin (excluding I_a) from the rainfall P. The potential retention S is the value that (F + I_a) would reach in a very long storm. According to the model:

$$Q = \frac{(P - 0.2S)^2}{P + 0.8S} \qquad (1)$$

which is the rainfall-runoff relation used in the SCS method (SCS, 1972). A transformation of S, the runoff curve number or hydrologic soil-cover complex number CN, was developed by SCS to facilitate with the calculations:

$$S = 25.4\left(\frac{1000}{CN} - 10\right)(mm) \qquad (2)$$

Substituting Equation 2 into Equation 1 gives the basic SCS relationship for estimating Q from P and CN, which has the advantage of having only one parameter since CN can readily be extracted from published tables, such as the following extract (Table 1), which contains only information relevant to this study:

Landuse/Landcover	**slope**	**Soil group**			
		A	**B**	**C**	**D**
Urban	≥3	96	96	96	96
	<3	93	93	93	93
Orchards	≥3	68	78	84	88
	<3	64	73	78	82
Irrigated	≥3	56	70	80	84
	<3	52	67	76	79
Bare	≥3	94	94	94	94
	<3	91	91	91	91
Matorral, (Xaló)		46	68	78	83
Matorral (Lesbos)		56	75	86	91
Maquis, sparse		46	68	78	83
Maquis, dense		40	60	69	76
Forest, decidious		36	52	62	69
Forest (Xaló)		40	60	69	76
Forest (Lesbos)		29	44	54	60

Table 1. SCS Curve numbers (after Ferrer et al., 1995)

As can be seen from the above sample table, the value of CN depends on factors such as landuse/cover, slope and soils which the SCS has divided into four groups according to their infiltration, retention and evaporation capacity. These factors control not only the amount of water that becomes runoff, but also the initial abstraction I_a, since they are closely related to the amount of interception, initial infiltration, surface depression storage, evaporation and transpiration.

In order to estimate runoff, the LULC maps were combined with the data of Table 1 to extract the necessary SCS CN. For the Xaló area, soil textural data, namely percentages of sand, silt and clay, from 16 point locations in the wider catchment area, were combined with lithological maps to extrapolate the point measurements over the area and produce three separate maps of percentage sand, silt and clay. These were then combined with the United States Department of Agriculture (USDA) soil textural triangle (Miller, 1996) to create a map of the four SCS soil groups. In the case of Lesbos, the previously mentioned 1:200000 textural soils data were employed. The resulting CN maps were then used along with Equations 1 and 2 and rainfall data to estimate runoff Q. As for the precipitation data, a uniform 200mm rainfall map, representing an event with a 10-year return period (Gisbert and Ibáñez, 2003), was used over Xaló. Due to the lack of such information for the island of Lesbos, rainfall values of a minimum of 100mm and a maximum of 200mm were distributed over the island according to the altitude of each pixel and the climatic zone to which it belonged, using the following equation:

$$P_X = 100 + 100\left(\frac{Z_X}{Z_{\max}}\right)w_i \qquad (3)$$

where P_X = precipitation at point **X** (mm)
Z_X = altitude at point **X** (m)
$Z_{\max}$ = maximum altitude over the island (m)
w_i = linear weight according to climatic zone

3.4 Erosion modelling

Thornes (1985, 1989) established a physical-based soil erosion model by combining sediment transport and vegetation protection in the following equation:

$$\mathrm{E} = kQ^m s^n e^{bVC} \qquad (4)$$

where E = erosion (mm)
k = soil erodibility coefficient
Q = overland flow (mm)
s = slope (%)
VC = vegetation cover (%)

The coefficients m and n have been described by a number of researchers. They vary according to different measurements: m changes between 0.91 to 2.07 and n from 0.24 and 1.67. Thornes (1976) suggested values of 2.0 for m and 1.67 for n. When modelling the competitive behaviour of vegetation and erosion, Thornes (1990) indicated that erosion is reduced exponentially in relation to the bare soil value by increased vegetation cover. The value b=-0.07 was used which is in accordance with the results of other researchers for a variety of environments (Drake et al., 2004, Symeonakis and Drake, 2004; Thornes, 1990).

Vegetation cover was then estimated using the NDVI estimate and the scaled NDVI or N*, which is equal to (Choudhury et al., 1994, Carlson et al., 1995):

$$N^* = \frac{NDVI - NDVI_0}{NDVI_S - NDVI_0} \qquad (5)$$

where $NDVI_S$ = the value of NDVI at 100% cover
$NDVI_S$ = the value of NDVI for bare soil

According to Carlson et al., (1995), the index N* is useful because it is relatively insensitive to viewing angle, sensor drift, and uncertainties in atmospheric corrections.

The Thornes erosion model was then applied for both dates of the two sites using the GRID module of ArcInfo v.8.1. By subtracting the runoff and erosion estimates of the earlier dates from their respective counterparts of the later dates, a qualitative comparison could be made and a possible increasing or decreasing trend could be identified. The results for the area of Xaló are shown in Figure 4 below:

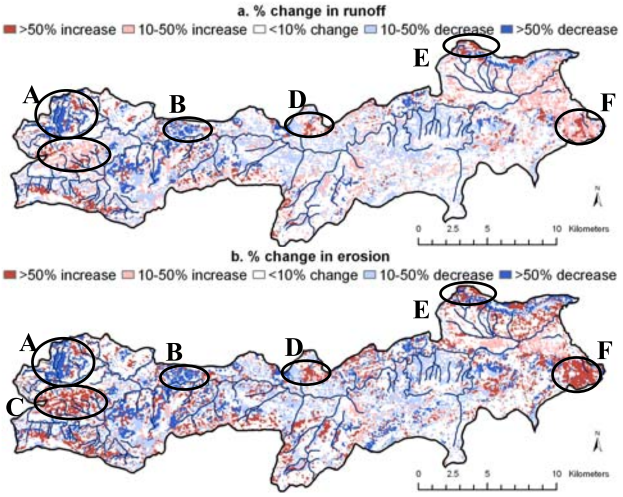

Figure 4: Xaló area (a) runoff estimate of 1978 subtracted from runoff estimate of 2000 and, (b) erosion estimate of 1978 subtracted from erosion estimate of 2000

Overall, there appears to be little change (<10%) in runoff and erosion throughout the Xaló catchment. Nevertheless, the figures also help identify areas where runoff rates and the erosion potential have both shown a considerable decrease of more than 50%, e.g. areas within eclipses A and B in Figure 4, possibly due to reforestation practices. Other areas have increased their runoff and erosion rates. This is mainly due to a significant documented change in landuse in the form of agricultural terrace abandonment (area C), or changes in land cover in the form of forest fires which are dominant in this part of Spain (García-Haro et al. 2001, areas D and E). Finally, in some areas downstream, new '*urbanizaciones*' or urbanized areas now dominate the landscape (area F), leading to an increase in runoff due to the replacement of the vegetated cover with paved roads or pavements. The actual effect this change might have on erosion rates and land degradation, is questionable since it greatly depends on the quality of the construction and the flood prevention engineering work carried out. Nevertheless, the 'visual' degradation of the environment in coastal Spain, with all its consequences on the sustainability and the livelihoods, is already quite pronounced and is thought to have long reached alarming rates.

The evolution of the runoff and erosion potential for the island of Lesbos is shown in Figure 5 below. Overall, there seems to be a greater amount of areas with >50% change in runoff and erosion rates than in the case of Xaló. So far the focus was on the western part of the island (through a series of MEDALUS…..199? projects) due to desertification that is taking place in this area. However, this study shows that, in relative terms, some areas in the south-eastern part of the island also appear to be at risk with increased amounts of runoff and erosion and therefore, prone to land degradation. The most threatened areas seem to be those where forest fires have destroyed either the coniferous and deciduous forests (areas A, B and C in figure 5) and are now covered by sparse trees and/or maquis. The areas where a small change or decrease in both runoff and erosion is more pronounced are those in the central part of the (areas D, E and between).

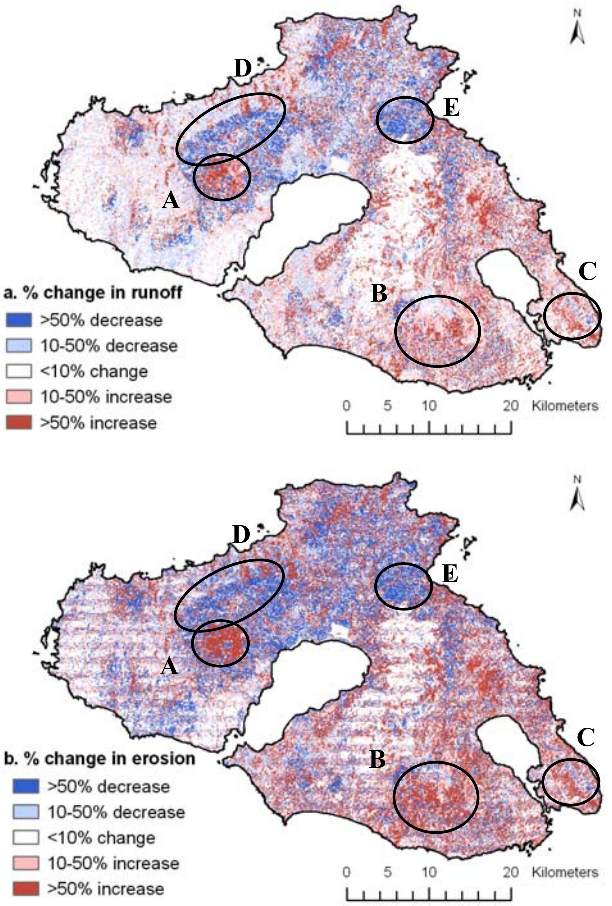

Figure 5: Lesbos (a) runoff estimate of 1975 subtracted from runoff estimate of 1999 and, (b) erosion estimate of 1975 subtracted from erosion estimate of 1999

4. CONCLUSIONS

This study has looked into the possibility of combining remotely-sensed with ancillary data, in order to study the potential relationship between LULC changes and land degradation in the Mediterranean region. This was achieved using two largely recognised indicators of degradation: surface runoff and sheetwash soil erosion. The results, produced using a high rainfall event, have identified areas where runoff and erosion figures are increasing as well as areas where these figures are less now than they were in the 1970s.

The methodology can be used in the decision-making process of the selection of areas that need mitigation measures to be adopted which will reduce or even reverse their degrading potential. Specifically, future work should focus on problems faced here with the specific datasets, such as: (a) the phenological difference of the MSS and the TM data as well as the calibration data, (b) the scarceness of the soil samples and the actual method of interpolating these over the area. More samples should be collected and a more reliable interpolation method such as the geostatistical method of kriging, should be applied, rather than extrapolating the point values over the lithological units. (c) With additional soils data, the soil erodibility factor of Thornes should be calculated to replace the k of USLE used in the case of Xaló. Finally, issues related to the propagation of errors from the combination of data derived from various sources and scales should also be tackled and are currently the focus of forthcoming research.

5. REFERENCES

Antolín, C., 1998. *El suelo como recurso natural en la Comunidad Valenciana. Vol II*. Consellería de Obras Públicas, Urbanismo y Transportes, Generalitat Valenciana. Valencia.

Belda, F., 1997. *Climatología y Teledetección en Zonas Forestales de la Provincia de Alicante. Aplicación en Zonas Incendiadas*. PhD Thesis, Facultat de Física, Departament de Termodinámica, Universitat de Valencia.

Brandt, J. and Thornes, J.B., 1996. *Mediterranean Desertification and Land Use*. Wiley, Chichester. p. 554

Camarasa-Belmonte, M., Bescós-Atín, A. and Sancho-Comíns, J., 1998. Evaluación del riesgo de erosión en relación con la dinámica ocupacional del suelo a partir de sistemas de información geográfica. In Gómez-Ortiz A. and F. Salvador-Franch (eds.) *Investigaciones Recientes de la Geomorfología Española*, Barcelona: pp. 579-592

Carlson, T. N., Capehart, W. J. and Gillies, R. R., 1995. A New Look at the Simplified Method for Remote Sensing of Daily Evapotranspiration. *Remote Sensing of Environment*, 54, pp. 161-167

Choudhury, B. J., Ahmed, N. U., Idso, S. B., Reginato, R. J. and Daughtry, C. S. T., 1994. Relations between Evaporation Coefficients and Vegetation Indices Studied by Model Simulations. *Remote Sensing of Environment*, 50, pp.1-17

Drake, N. A. and Vafeidis, A., 2004. A review of European Union funded research into the monitoring and mapping of Mediterranean desertification. *Advances in Environmental Monitoring and Modelling*, 1(4), pp. 1-51

Drake, N. A., Zhang, X., Symeonakis, E., Wooster, M., Patterson, G. and Bryant, R., 2004. Near real-time modelling of regional scale soil erosion using AVHRR and Meteosat data: A tool for monitoring the impact of sediment yield on the biodiversity of Lake Tanganyika, in *Spatial Modelling of the Terrestrial Environment*, by Kelly, R., N. Drake and S. Barr (eds.). Wiley, Chichester. p. 290

Ferrer, M., Rodríguez, J. and Estrela, T., 1995. Generación automática del número de curva con SIG. *Ingeniería del Agua*, 2(4), pp. 43-58

Garcia-Haro, F.J, Gilabert, M.A. and Melia, J., 2001. Monitoring fire-affected areas using Thematic Mapper data. *International Journal of Remote Sensing*, 22(4), pp.533-549

Kosmas, C., Kirkby, M. and Geeson, N., 1999. *Manual on key indicators of desertification and mapping environmentally sensitive areas to desertification. European Commission*, EUR 18882.

Kosmas, C., Danalatos, N.G. and Gerontidis, St., 2000a. The effect of land parameters on vegetation performance and degree of erosion under Mediterranean conditions. *Catena*, 40, pp. 3-17

Kosmas, C., Gerontidis, St. and Marathianou, M., 2000b. The effect of land use change on soils and vegetation over various lithological formations on Lesvos. *Catena*, 40, pp. 51-68

Krugmann, H., 1996. Toward Improved Indicators to Measure Desertification and Monitor the Implementation of the Desertification Convention. http://web.idrc.ca/en/ev-30796-201-1-DO_TOPIC.html (accessed 14 Apr. 2004)

Loumou A., Giourga, Ch., Dimitrakopoulos, P. and Koukoulas, S., 2000. P. Tourism contribution to agro-ecosystems conservation; the case of Lesbos island, Greece. *Environmental Management*, 26(4), pp. 363-370

Maidment, D. R. (ed.). 1993. *Handbook of Hydrology*. McGraw-Hill, New York.

Miller, G. T., 1996. *Living in the Environment; Principles, Connections, and Solutions*. Wandsworth, p. 727

Rubio, J. L. and Bochet, E., 1998. Desertification Indicators as Diagnosis Criteria for Desertification Risk Assessment in Europe, *Journal of Arid Environments*, 39, pp.113-120

Sharma, K. D., 1998. The Hydrological Indicators of Desertification, *Journal of Arid Environments*, 39, p.121-132

Soil Conservation Service, 1972. SC*S National Engineering Handbook, Section 4: Hydrology*, USDA.

Symeonakis, E. 2001. *Soil Erosion Modelling over Sub-Saharan Africa using Remote Sensing and Geographical Information Systems*. PhD Thesis, King's College, University of London. p. 379

Symeonakis, E., Calvo, A. and Arnau, E., 2003. A study of the relationship between landuse changes and land degradation in SE Spain using remote sensing and GIS. Proceedings of the International Conference of GIS and Remote Sensing in Hydrology, Water Resources and Environment, Three Gorges Dam site, China, 16-19 September 2003.

Symeonakis, E. and N. Drake. 2004. Monitoring desertification and land degradation over sub-Saharan Africa. International Journal of Remote Sensing, 25, 3: 573-592.

Symeonakis, E., Calvo, A. and Arnau, E., in press. A landuse change and land degradation study in a Mediterranean catchment using Remote Sensing and GIS. *Environmental Management.*

Thornes, J. B. 1976. Semi-Arid Erosional Systems. Geographical Papers No 7. London School of Economics and Political Science.

Thornes, J. B. 1985. The ecology of erosion. Geography, 70: 222-234.

Thornes, J.B., 1989, Erosional equilibria under grazing. In Bintliff, J., Davidson, D., and Grant, E. (eds), Conceptual Issues in Environmental Archaeology, (University Press: Edinburgh), 193-210.

Thornes, J. B. 1990. The interaction of erosional and vegetation dynamics in land degradation: spatial outcomes. In J. B. Thornes (ed.) Vegetation and Erosion. John Wiley and Sons, Chichester: 41-53.

Thornes, J., 1996. Desertification in the Mediterranean. In: Brandt, J., Thornes, J. (eds.), *Mediterranean Desertification and Land Use*. Wiley, Chichester, pp. 1–12.

Viedma, O. 1999. Caracterización Espectral y Espacial de los Incendios Forestales Mediante Técnicas de Teledetección. PhD Thesis, Departamento de Geografía, Universidad de Alcalá de Henares.

UNEP, 1990. Global Assessment of Land Degradation/Desertification – GAP II. Desertification Control Bulletin, 18, 24-25.

Viedma, O. and J. Melia. 1999. Monitoring temporal changes in the spatial patterns of a Mediterranean shrubland using Landsat registered images. Diversity and distributions. Special Issue: Global Change Effects on Landscape and Regional Patterns of Plant Diversity, 5, 6: 275-293.

6. ACKNOWLEDGEMENTS

This research forms part of a Research Training Project financed by the European Community with Contract no: EVK2-CT-2001-50014. The authors are also grateful to Dr Vicente Pons, Departamento de Biología Vegetal, Universitat de València, for providing the Xaló soil samples data.